Water quality modeling of large reservoirs in semi-arid regions under climate change – Example Lake Nasser (Egypt)

Von der
Fakultät Architektur, Bauingenieurwesen und Umweltwissenschaften
der Technischen Universität Carolo-Wilhelmina
zu Braunschweig

zur Erlangung des Grades eines
Doktoringenieurs (Dr.-Ing.)
genehmigte

Dissertation

von
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Berichtersttater	Prof. Dr.-Ing. G. Meon
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Dedicated to
my great country, Egypt
the memory of my beloved parents and sister
my kind wife and beautiful daughters
Maryam and Nour

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Abstract

Maintaining or improving the water quality of reservoirs is of particular concern. In fact, it is considered now as one of the key factors in the operation and water quality management of reservoirs, particularly in regions that are exposed to high variations of climatology, like Egypt. Egypt is extremely dependent on the River Nile; the country hardly has any other fresh water resources. Full control of the Nile water discharge was achieved in the 1970s after the construction of the Aswan High Dam (AHD). As a result, the AHD reservoir was formed. It is considered as one of the largest man-made lakes in the world. Egypt has realized the importance of water quality of the AHD reservoir, as this reservoir is almost the sole source of fresh water in the country. The length of the submerged area is at present 500 km, of which 350 km are within the Egyptian territory, and it is known as Lake Nasser, while the 150 km stretch which lies in the northern most part of the Sudan is known as Lake Nubia.

In this work, a hydrodynamic and water quality model was developed for Lake Nubia based on a two-dimensional, laterally averaged and finite difference hydrodynamic and water quality code, CE-QUAL-W2. The model was calibrated and verified using data which were measured in the years of 2006 and 2007 during low flood periods, respectively. Measurements during the flood season are not available. The results of the presented model show a good agreement with the observed hydrodynamic and water quality records.

Two water quality indices (WQIs), NSF WQI and CCME WQI, have been developed to assess the state of water quality in the investigated case study, Lake Nubia, during the first low flood period of January 2006. The CCME WQI has been modified to use the Egyptian standards (objectives) of raw water. Moreover, another two trophic status indices, Carlson TSI and LAWA TI, have been developed to evaluate the trophic status of Lake Nubia during the same period of January 2006. Results of the previously developed hydrodynamic and water quality model for Lake Nubia were used to validate the model. According to the developed water quality indices results, Lake Nubia has a good water quality state during the low flood period. The modified CCME WQI (based on measured data) indicates that the Lake Nubia water quality state is excellent according to the Egyptian standards of water quality for surface waterways. Results of the applied trophic status indices show that the Lake Nubia trophic status is eutrophic during the studied period.

The effect of the global climate change on the hydrodynamic and water quality characteristics of Lake Nubia was conducted for the 21st century. To do that, the outputs of eleven global climate models for two global emissions scenarios combined with hydrological modeling were used. A theoretical process algorithm has been simplified, further developed and calibrated to modify the initial conditions of dissolved oxygen due to the global climate change effects. A sensitivity analysis has been conducted by using each of the predicted air temperature and inflow data separately in the model in order to investigate its effect on the characteristics of the hydrodynamic and water quality. Three hydrodynamic characteristics of the reservoir were investigated with respect to the climate change: water surface levels, evaporation water losses and thermal structure. In addition, eight water quality characteristics of the reservoir were investigated with respect to the climate change: dissolved oxygen, chlorophyll-a, ortho-phosphate, nitrate-nitrite, ammonium, total dissolved solids, total suspended solids and potential of hydrogen (pH). Moreover, the climate change effects on the water quality and trophic status indices have been studied. The results of the climate change study show partially significant impacts on the examined hydrodynamic and water quality characteristics, while the water quality and trophic status indices are slightly affected by the climate change scenarios.

Kurzfassung

Die Effekte des globalen Klimawandels bedingt durch die Emission von Treibhausgasen stehen derzeit auf der aktuellen wissenschaftlichen und politischen Agenda. Weltweite Beobachtungen zeigen eine Temperaturzunahme in Seen und Flüssen, mit Effekten auf die thermale und chemische Struktur.

Derzeitige Bemühungen zielen darauf ab, die Wasserqualität auf dem jetzigen Stand beizubehalten oder zu verbessern. Der Betrieb von Reservoiren und deren Wasserqualitätsmanagement ist ein Schlüsselfaktor in Ländern mit hoher klimatischer Variabilität, wie z.B. Ägypten. Das Land ist in hohem Maße vom Fluss Nil abhängig, weitere Frischwasserquellen stehen dem Land kaum zur Verfügung. Durch den Bau des Assuan-Staudamms in den 1970er Jahren wurde eine komplette Kontrolle des Nilabflusses erreicht. Durch das Aufstauen des Nils entstand eines der größten, durch Menschenhand geschaffenen Reservoire der Erde. Das erklärte Ziel Ägyptens ist es, eine gleichbleibend gute Wasserqualität zu gewährleisten, da das Reservoir die einzige nennenswerte Frischwasserquelle des Landes ist. Die Längsausdehnung des Stausees beträgt 500 km, wobei 350 km auf ägyptischem Gebiet liegen (Nasser-See) und 150 km auf sudanesischem Gebiet liegen (Nubia-See).

Im Mittelpunkt dieser Arbeit steht die Analyse des Klimawandeleinflusses auf die Hydrodynamik und Wasserqualität des Nubia-Sees. Die Untersuchung wurde für den Zeitraum des 21. Jahrhunderts, unter Verwendung der Ergebnisse aus elf globalen Klimamodellen für zwei Emissionsszenarien, inklusive der Kopplung mit hydrologischen Modellen, durchgeführt. Die Analyse der Hydromechanik und Charakteristika der Wasserqualität in einer Wassersäule basiert auf einem zweidimensionalen Finite-Differenzen-Modell (CE-QUAL-W2). Sowohl die Kalibrierung, als auch die Validierung beruhen auf Messserien der Jahre 2006 und 2007. Es wurden drei hydrodynamischen Eigenschaften des Reservoirs, die als Indikator zur Ausweisung eines Klimawandels dienen, untersucht: Wasserstände, Wasserverluste durch Evaporation und thermische Schichtung. Folgende hydrochemische Parameter wurden ebenfalls in Abhängigkeit des Klimawandels untersucht: gelöster Sauerstoff, pH-Wert, Chlorophyll-A, Ortho-Phosphate, Nitrat / Nitrit, Ammonium, sowie Sedimentfrachten und Schwebstoffe. Die Ergebnisse zeigen einen partiell signifikanten Einfluss des globalen Klimawandels in der Hydrodynamik und der Wasserqualität des Reservoirs.

Die Auswirkungen des globalen Klimawandels auf zwei Wasserqualitätsindikatoren und zwei Trophieindikatoren im Nubia-See wurden in weiteren Untersuchungen belegt. Diese Ergebnisse

weisen einen geringen Einfluss des globalen Klimawandels in Bezug auf Wasserqualität und Trophiegrad auf.

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Abbreviation

A2	A2 Emission Scenario
AHD	Aswan High Dam
AME	Absolute Mean Error
AMSL	Above Mean Sea Level
AR4	IPCC Fourth Assessment Report
B1	B1 Emission Scenario
BCM	Billion Cubic Meter
CCME	Canadian Council of Ministers of the Environment
Chl-a	Chlorophyll-a (mg/L)
DJF	December, January and February
DO	Dissolved Oxygen (mg/L)
DO_{sat}	Dissolved Oxygen Saturation concentration (mg/L)
EEAA	Egyptian Environmental Affairs Agency
FC	Fecal Coliform (Cell Numbers/100 mL)
GCC	Global Climate Change
GCM	General Circulation Model
GHG	Global greenhouse gas
GIS	Geographic Information System
IPCC	Intergovernmental Panel on Climate Change
LAWA	Länderarbeitsgemeinschaft Wasser
LNDFC	Lake Nasser Flood and Drought Control project
MWRI	Egyptian Ministry of Water Resources and Irrigation
NH_4	Ammonium (mg/L)
NO_2	Nitrite (mg/L)
NO_3	Nitrate (mg/L)
NSF	National Sanitation Foundation
OECD	Organization for Economic Cooperation and Development
PCMDI	Program for Climate Model Diagnosis and Intercomparison
pH	Potential of Hydrogen (unit)
PO_4	Ortho-Phosphate (mg/L)
RMS	Root Mean Square Error
SD	Secchi Depth (m)
SRES	Special Report on Emission Scenarios
St. n	Station Number n
T	Temperature (°C)
TDS	Total Dissolved Solids (mg/L)
TI	Total Index
TIN	Triangulated Irregular Network
TN	Total Nitrogen (mg/L)

TP	Total Phosphorus (mg/L)
TSI	Trophic Status Index
TSS	Total Suspended Solids (mg/L)
UNEP	United Nations Environment Programme
USACE	United States Army Corps of Engineers
USEPA	United States *Environmental Protection Agency*
USGS	U.S. Geological Survey
WHO	World Health Organization
WQI	Water Quality Index

1. Introduction

No doubt, water is essential for maintaining life on Earth. In pictures taken by astronauts circling the Earth, we see the globe covered mostly by a rich, blue color signifying the presence of huge quantities of water. In fact, this picture is deceiving (Rast and Straskraba, 2000) because more than 97.5% of the Earth's water is stored in the oceans, leaving only about 2.5% of fresh water. Additionally, of that 2.5% only 0.4% is readily available on the surface. Figure 1.1 shows that natural lakes in world, on which hundreds of millions of people depend for their drinking water, contain more than 50% of the total amount of readily available fresh water.

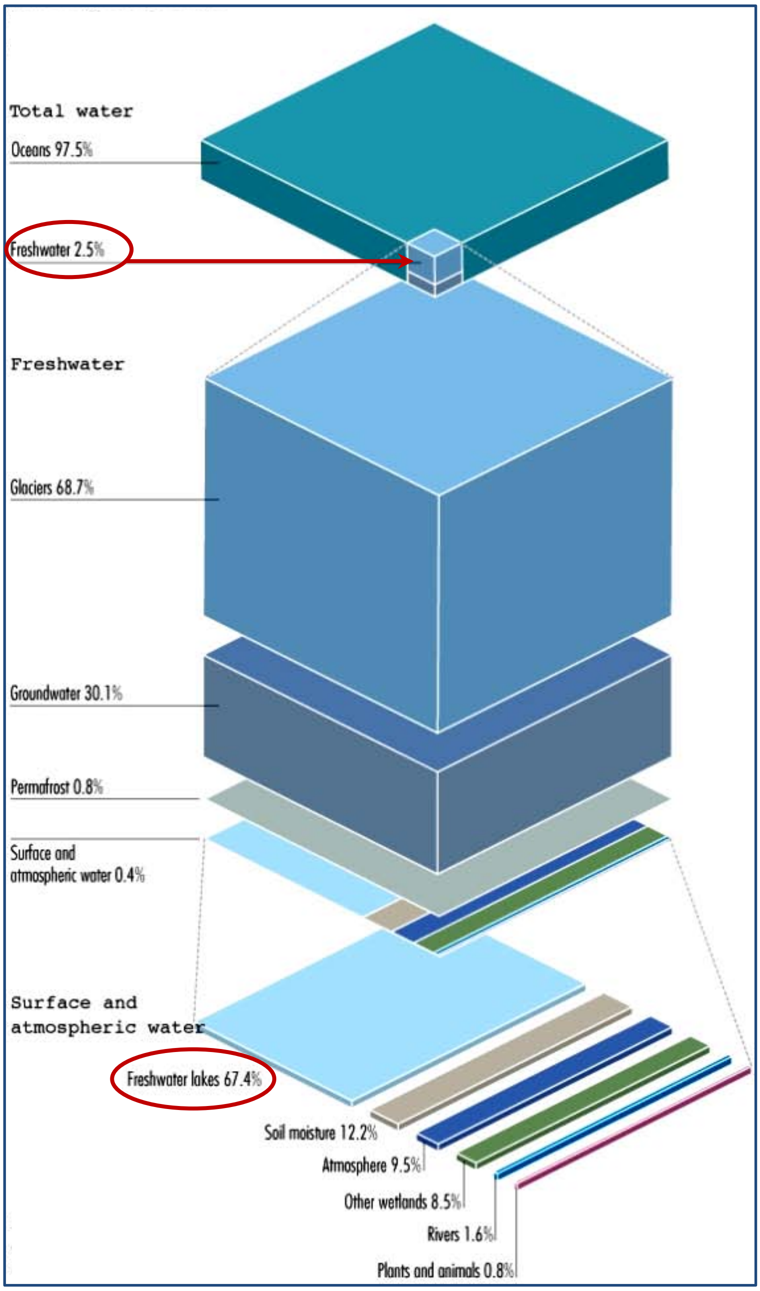

Figure 1.1: Distribution of global water, modified after UNEP (2007).

Over the last 50 years, the world population has almost tripled, while the use of water during the same period has increased more than the increase in the world population (three folds). The exploitation of lands, bodies of water and forests resulting from development has increased dramatically. This has led to a number of pronounced problems in lakes and reservoirs (Dinar et al., 1995). Africa's Lake Chad has shrunk dramatically in the last four decades due to increased water use and low rainfall, and it is destined to become a "puddle" (Adenle, 2002). Moreover, in 1987, the World Commission on Environment and Development warned in the final report (Our Common Future (WCED, 1987)) that water was being polluted and water supplies were overused in many parts of the world. Observational evidence from all continents and most oceans shows that many natural systems are being affected by regional climate changes, particularly temperature increases (IPCC, 2007a).

In Egypt, the improvement of the environmental quality is one of the most important challenges (MWRI 2005). A rapid deterioration of the Egyptian water resources has been occurred due to the increase of population, industrial and agricultural activities. Egypt is one of the most vulnerable countries to the potential impacts and risks of climate change. The sectors of water resources, agricultural resources and food security, coastal resources, tourism and health are all highly vulnerable with serious socioeconomic implications (EEAA, 1999; 2010). More than 95% of the water budget of Egypt is received from the River Nile, which is generated outside Egypt's territory. Numerous studies showed that the River Nile is very sensitive to temperature and precipitation changes (Riebsame et al., 1995). Considering above conditions, scientific-based modeling tools for the prediction of water resources with regard to water quantity and water quality aspects have to be developed and increasingly carefully applied. These tasks need to be done in a more sophisticated and accurate way, than presently performed in the academic and engineering practice in many semi-arid countries. In particular, the developing of water quality models for different Egyptian water resources is essential. Such models can be used to manage and forecast water resources responses under varying conditions.

1.1 Objectives and approaches of this research work

The main objectives of this research work are to develop a hydrodynamic and water quality model to investigate the water quality aspects of reservoirs in semi-arid regions with a focus on an Egyptian case study, to use the case study model to estimate the water quality and trophic status indices at different scenarios (in the present and future) and use the case study model to investigate the global climate change impacts. To achieve these objectives, the research approaches are divided into three stages (Figure 1.2):

- Stage I: Hydrodynamic and water quality modeling of lakes and reservoirs.

This stage includes:

1. State-of –the-art of water quality characteristics and issues.
2. State-of –the-art of hydrodynamics and water quality theoretical modeling approaches.
3. State-of –the-art of the available hydrodynamic and water quality modeling systems (codes).
4. Choosing a powerful hydrodynamic and water quality modeling system (code).

5. Choosing the vital investigated case study in Egypt to collect the required input data for the modeling system.
6. Developing a hydrodynamic and water quality model for the investigated case study.
7. Validating (calibrating and verifying) the presented model.

- Stage II: Water quality and trophic status assessment approaches (indices).

This stage includes:

1. State-of –the-art of numerous water quality and trophic status assessment approaches (indices) for surface freshwater resources.
2. Implementing two different water quality indices (WQIs) for the investigated case study (comparative study).
3. Implementing two different trophic status indices (TSIs) for the investigated case study (comparative study).
4. Verifying the presented hydrodynamic and water quality model of the investigated case study to estimate four different water quality and trophic status indices.

- Stage III: Impacts of the global climate change.

This stage includes:

1. State-of –the-art of global climate change theory and its effects on water resources.
2. Estimating the future climate changes for the investigated case study.
3. Developing the presented hydrodynamic and water quality model to estimate the climate change impacts on the hydrodynamic and water quality characteristics of the investigated case study.
4. Implementing the presented hydrodynamic and water quality model to estimate the climate change impacts on the different four water quality and trophic status indices of the investigated case study.

1.2 Impacts of this research work

It was addressed in the literature that there are no previous hydrodynamic and water quality models have been developed for the selected Egyptian case study (Aswan High Dam, or Lake Nasser, reservoir), although this huge reservoir is very vital to Egypt; it provides more than 95 percent of the Egyptian water budget. This reservoir is threatened by different water pollution hazards due to land use projects in Sudan and south of Egypt. Moreover, the reservoir inflow and its hydrodynamic and water quality characteristics will be affected by the global climate change. Therefore, this research work will contribute to the following aspects:

- Water quality management of water resources by using water quality models as management tools to evaluate different management strategies.
- The impacts of the global climate change on water resources.
- Developments of water quality monitoring programs according to input data of water quality models.

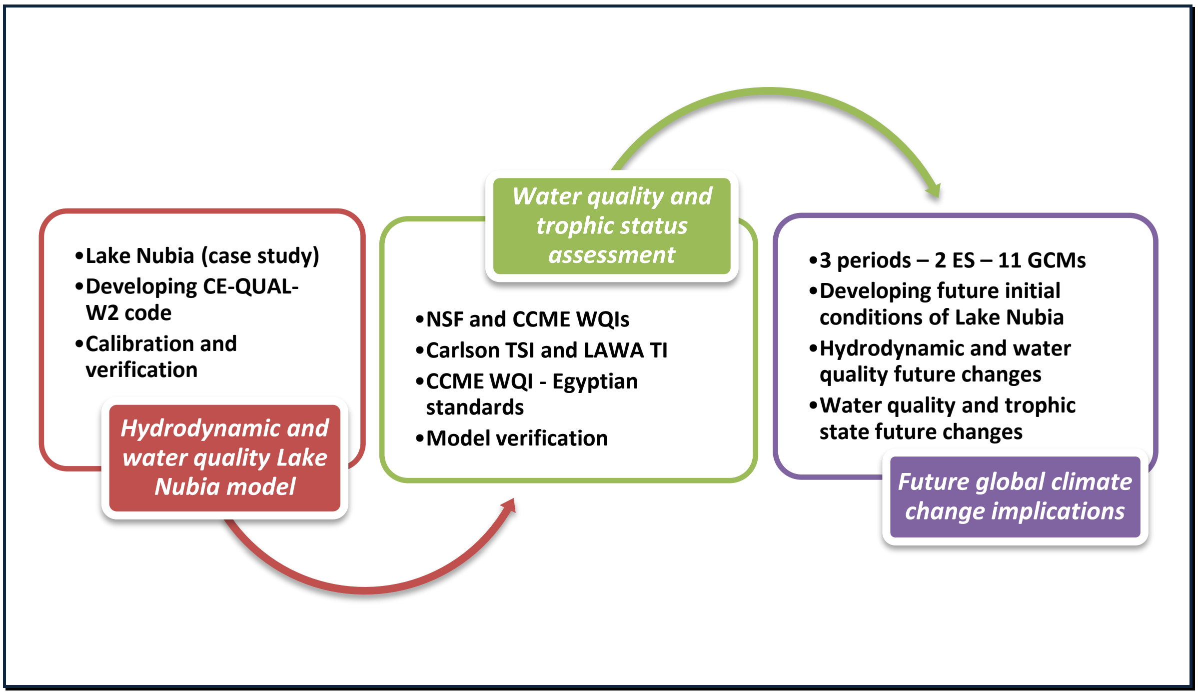

Figure 1.2: Approaches of the current research.

1.3 Thesis outline

The thesis is divided into seven chapters. In the following, the chapters are briefly discribed:

- **Chapter 1**: Introduction to the thesis.
- **Chapter 2**: presents a review on lake and reservoir water quality characteristics, hydrodynamics, theoretical modeling approaches and the available hydrodynamic and water quality modeling systems (codes).
- **Chapter 3**: includes the selected hydrodynamic and water quality code description, selected investigated area characterization and the collection of the required input data.
- **Chapter 4**: introduces the implementation of the selected hydrodynamic and water quality modeling system (code) for the selected investigated area. Calibration and verification of the proposed model are also presented.
- **Chapter 5**: The implementation of four different water quality and trophic status indices for the investigated case study is presented in this chapter. Moreover, this chapter presents the verifying of the proposed hydrodynamic and water quality model to estimate the four different water quality and trophic status indices.
- **Chapter 6**: Estimation of the future climate changes for the investigated case study is introduced in this chapter. Implementation of the proposed hydrodynamic and water quality model to estimate the climate change impacts on the hydrodynamic and water quality characteristics of the investigated case study is also presented. Moreover, the development of the proposed hydrodynamic and water quality model to estimate the climate change impacts on the different four water quality and trophic status indices is included.

- **Chapter 7**: Summarizes the main conclusions and recommendations of the current research work.

2. State of the art

2.1 Preface

The water cycle (also known as the hydrologic cycle), as defined by Ji (2008), represents the movement and endless recycling of water between the atmosphere, the land surface and the ground (Figure 2.1). The water cycle begins with water evaporation from the earth's water surface, soil and plants. Once in the air, the water vapor is transported by winds and may later condense into clouds. A portion of the water vapor falls to the ground as precipitation in the form of rain or snow. As the precipitation returns water to the land surface, a portion of it seeps into the ground and becomes groundwater. The remaining portion which does not infiltrate the soil but flows over the surface of the ground to a stream is called surface runoff. The water flowing through the ground can also return to the surface to supply water to rivers and lakes. All of the land that eventually drains to a common river or lake is considered to be in the same watershed. By a network of streams, the water that is not evaporated back into the atmosphere eventually reaches the oceans. Therefore, land use activities in a watershed can affect the water quality of surface waters. To accurately estimate pollution loadings to a surface water system, the water cycle of the watershed must be considered accordingly (Ji, 2008).

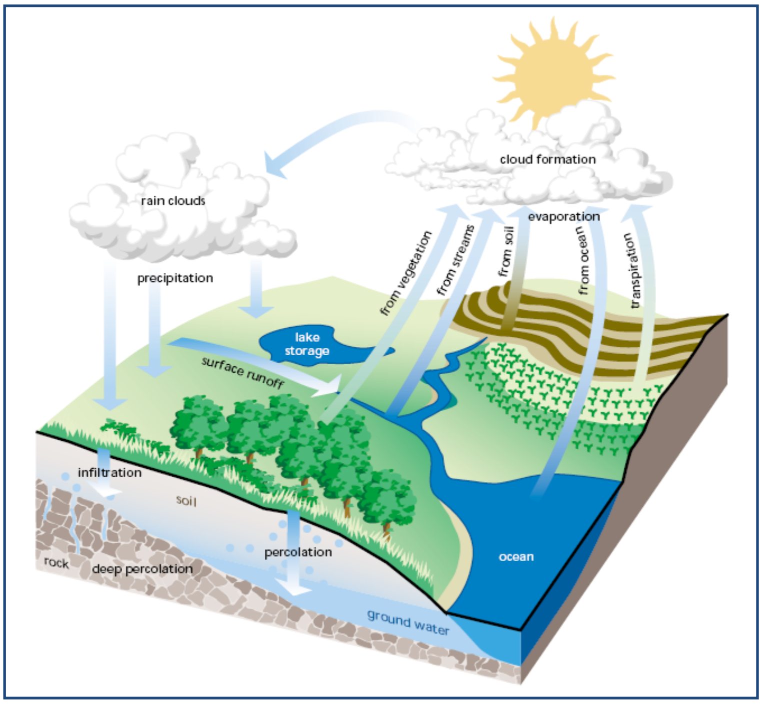

Figure 2.1: The Hydrologic Cycle (FISRWG, 1998).

The present chapter provides a literature review of lakes and reservoirs (uses, origin, morphological, physical, chemical and biological characteristics and water quality issues) and their water quality modeling (modeling, hydrodynamics, theoretical approach and available water quality models).

2.2 Lakes and reservoirs

Lakes and reservoirs are bodies of surface water that fill some basins, or hollows, in the earth's surface (Martin et al., 1999). Lakes are commonly referred to as a natural water bodies formed by geological processes, such as receding glaciers, volcanoes and earthquakes (Ji, 2008). While reservoirs are those water bodies formed or modified by human activity for specific purposes, in order to provide a reliable and controllable resource. Their main uses include (Thornton et al., 1996):

- Drinking and municipal water supply.
- Industrial and cooling water supply.
- Power generation.
- Agricultural irrigation.
- River regulation and flood control.
- Commercial and recreational fisheries.
- Body contact recreation, boating and other aesthetic recreational uses.
- Navigation.
- Waste disposal (in some situations).

There are many different types of lakes and reservoirs; and they serve many functions and purposes. Some lakes are ephemeral, such as floodplain lakes of large rivers. Others may be located in dry regions and are saline like Lake Qarun in Egypt or the Great Salt Lake in America. Small saltwater lakes in coastal zones are essential for production of certain fisheries. Some lakes are recharged almost totally from groundwater inflow, while others, like Lake Chad in Africa or small lakes in floodplains or delta areas in Asia, serve to recharge groundwater supplies. Depending on size, depth, climate and flow regime, lakes differ in vulnerability to pollution (Dinar et al., 1995).

Most reservoirs were built for a single purpose; storage of a certain quantity of water. With increasing environmental degradation and multiple uses of reservoirs, water quality has become an issue of great concern. Drinking water supply has the greatest requirements for high water quality. In addition, some technical processes require water that possesses specific quality limits, and fishes cannot thrive and remain edible for humans in highly polluted waters. Recreational values, another recent reservoir use, also necessitate relatively clean water.

2.2.1 Formation of lakes and reservoirs

Lakes are water-bodies formed by nature whereas reservoirs are artificial water bodies constructed by humans, either by damming a flowing river or by diverting water from a river to an artificial basin (impoundment). Many characteristics of lakes and reservoirs are a function of the way in which they were formed and the human uses.

2.2.1.1 Natural lakes

Most natural lakes were formed by catastrophic events, while others were formed by gradual events (Cohen, 2003; Dodson, 2005; Löffler, 2004; Martin et al., 1999; NALMS et al., 2001).

Catastrophic events:

1- Glacial lakes,

The erosional and depositional activity of glaciers was the most important agent of lake formation. Many lakes were formed from the outwash morainal deposits at the retreating edges of continental glaciers or from melted blocks of ice buried in this morainal debris, Kettle Lakes, as can be shown in Figure 2.2. Lakes of glacial origin make up about 48% of the area and 22% of the volume of all lakes on the earth. Almost half of the world's large lakes more than 500 km^2 are glacial, and these water bodies account for nearly a third of the area of large lakes (Cohen, 2003). Most glacially formed lakes lie north of 40^{o}N (e.g. in Canada, Russia, Scandinavia).

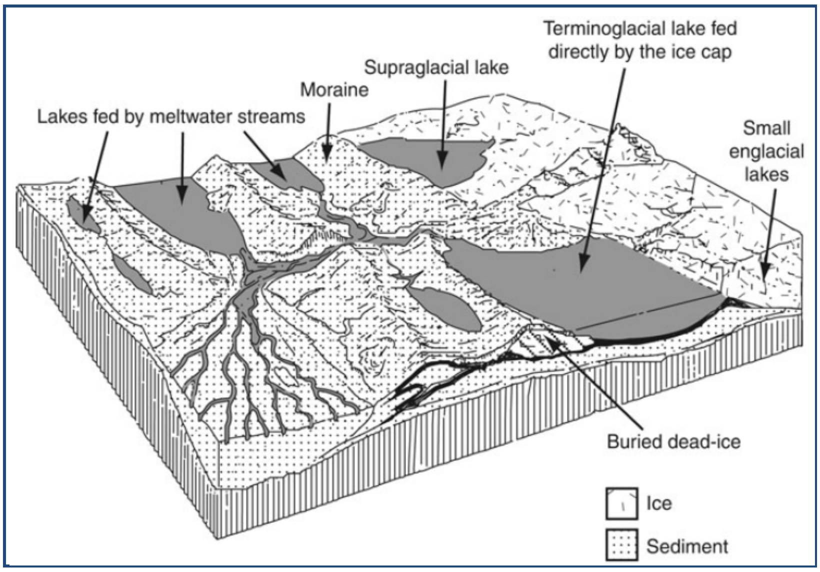

Figure 2.2: Schematic representation of a variety of glacial settings for lake formation (Cohen, 2003).

2- Tectonic lakes,

Tectonic processes are responsible for the evolution of a vast number of lakes, including the largest, deepest and oldest lakes on the planet. Although they account for a smaller proportion of total lake area (40%) than glacial lakes, their greater average depth makes them by far the largest class of lakes by volume (75%) (Cohen, 2003). Continent-sized pieces of the Earth's surface (tectonic plates) move slowly relative to one another. Where two plates meet, the surface of the Earth will sometimes break (fault or rift), with one plate moving downward relative to the other. The African rift lakes, such as Lake Tanganyika, occur along just such a major fault line. The basin created by the broken crust is called *graben* and can host a lake. For example, Lake Baikal in Russia is a 1741-meter-deep lake on top of more than 3000 meters of lake sediments (Dodson, 2005).

3- Volcanic lakes,

Volcanic activity produces craters, also called calderas, which, in some cases, hold water. An older and smaller volcanic crater lake is the one that hosts the Lago di Monterosi in central Italy (Figure 2.3). This lake is about 25.000 years old, in a crater just a few meters deep and mostly filled with wind-blown sediments. If the volcano is still at all active, the crater lake will probably have unusual and acidic water chemistry (Dodson, 2005), Figure 2.4. Volcanic lakes as a whole account for <1% of global lake area (Cohen, 2003).

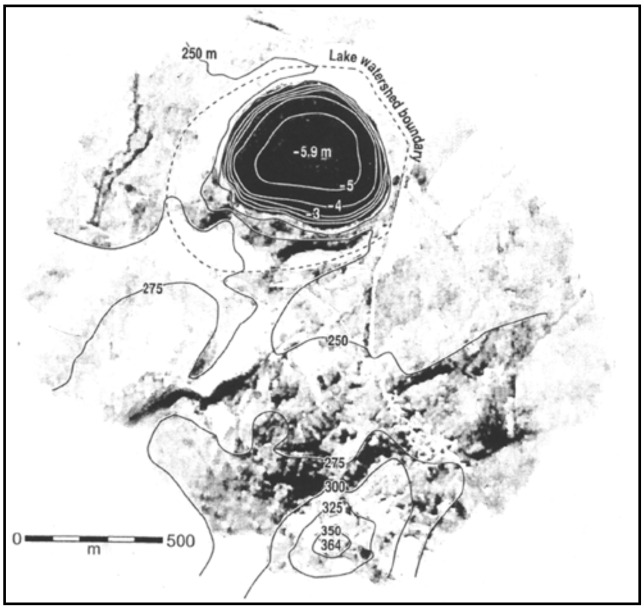

Figure 2.3: Topographic map of Lago di Monterosi, Italy (Dodson, 2005).

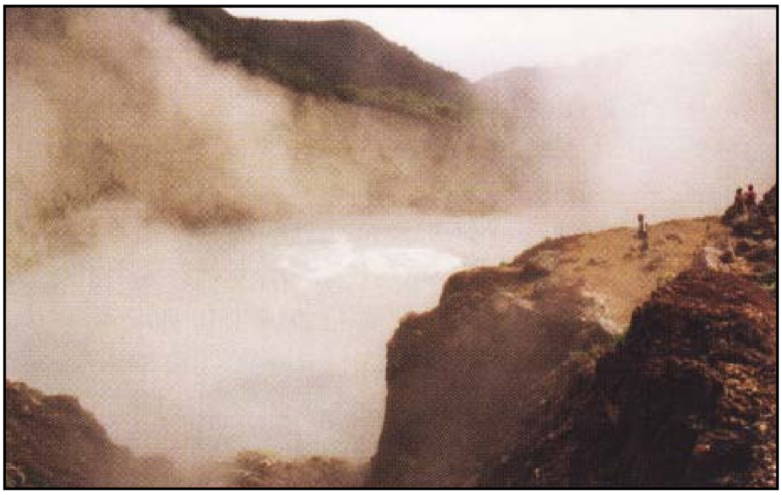

Figure 2.4: Boiling lake in an active caldera on the Caribbean island of Dominica (Dodson, 2005).

4- Lakes formed by meteorites,

Impacts of rocks from the sky also produce deep basins (impact craters). The impact of a stony meteorite in northern Argentina created several basins that filled with water and became lakes (Dodson, 2005).

5- Lakes formed by landslides,

Sudden movements of large quantities of unconsolidated material in the form of landslides into the floors of stream valleys can cause dams and create lakes, often of very large size (Wetzel, 2001).

Gradual events:

1- Coastal lakes,

Coastal lakes (or shoreline lakes) often form along irregularities in the shore line of the sea or large lakes. Long-shore currents deposit sediments in bars or spits that eventually isolate a fresh or brackish-water lake, such as for the coastal lakes in Egypt (Figure 2.5). Coastal lakes formed by such blockages comprise about 60,000 km^2, about 2.3% of total lake area, most such lakes are shallow; the mean maximum depth of the largest 10 coastal lakes (> 500km^2) is only 6 m (Cohen, 2003).

Figure 2.5: Coastal lakes, Egypt (MWRI, 2005).

2- Solution lakes,

Solution lakes are formed when solution caves collapse. They are normally called *Karst lakes and* are very common in limestone regions (e.g. Lake Ohrid, Yugoslavia).

3- Fluvial lakes,

Stream deposition or erosion can isolate depressions to form lakes. *Plunge-pool lakes* form from erosion below former waterfalls, such as the Grand Coulee Lake, USA. *Oxbow lakes* are formed with the cut-off of former river channel. Fluvial lakes make up 8% of the world's total lake area, but only about 0.3% of total world lake volume (Cohen, 2003).

4- Lakes formed by wind erosion,
Wind erosion can form shallow depressions that often contain water temporarily or seasonally, such as *dune lakes* in sandy areas and *playa lakes* in arid regions (e.g. the arid regions of USA and Xinjiang Province in western China) (Martin et al., 1999).

There are many other types of natural lake basins. For example, Hutchinson and Edmondson (1957) listed 76 types of lakes based on their geomorphologic characteristics. However, the above classification provides some indications of the diversity of lakes that can occur.

2.2.1.2 Man-made lakes (reservoirs)
Reservoirs – man-made lakes – are artificial lakes that were created by man for particular purposes (Straškraba and Tundisi, 1999). This creation makes reservoirs different in many aspects than lakes. Nature lakes fill depressions, whereas reservoirs usually fill a river valley that has been dammed by a wall. As is true of lakes, reservoirs are rather variable; there is no uniformity of location, size, or shape. Reservoirs are usually found in areas of water scarcity, or where a controlled water facility was necessary. Small reservoirs were first constructed some 4,000 years ago in Egypt (Figure 2.6), China and Mesopotamia, primarily to supply drinking water and for irrigation purposes (Rast and Straskraba, 2000).

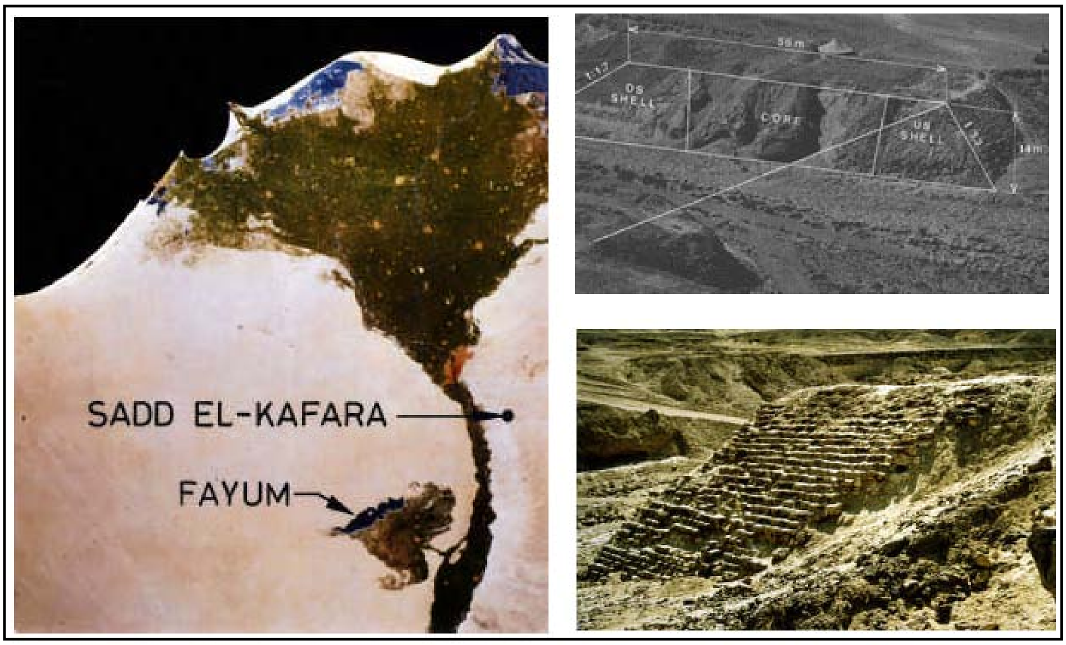

Figure 2.6: Sadd El-kafara, Egypt, the world's oldest large dam. The dam was built around 2600 BC and was 14 m high and 113 m along the crest. The figure shows the position, the ruins and the upstream view of the dam (Cracking Dams website; Garbrecht, 1983; 2006; Schnitter, 1994).

Types of reservoir systems:
The term 'reservoir' includes several types of constructed water-bodies and/or water storage facilities. These types are (Rast and Straskraba, 2000):

1- Valley reservoirs
Valley reservoirs are created by constructing a barrier (dam) perpendicular to a flowing river.

2- Off-river storage reservoirs

Off-river storage reservoirs are created by constructing an enclosure parallel to a river. Subsequently, these reservoirs are supplied with water either by gravity or by pumping from the river. The latter reservoirs are sometimes also called embankment or bounded reservoirs, which controls inflows and outflows to and from one or more rivers.

In addition to single reservoirs, reservoir systems also exist and include:

1- Cascade reservoirs

Cascade reservoirs are consisting of a series of reservoirs constructed along a single river.

2- Inter-basin transfer schemes

Inter-basin transfer schemes are designed to move water through a series of reservoirs, tunnels and/or canals from one drainage basin to another (Figure 2.7).

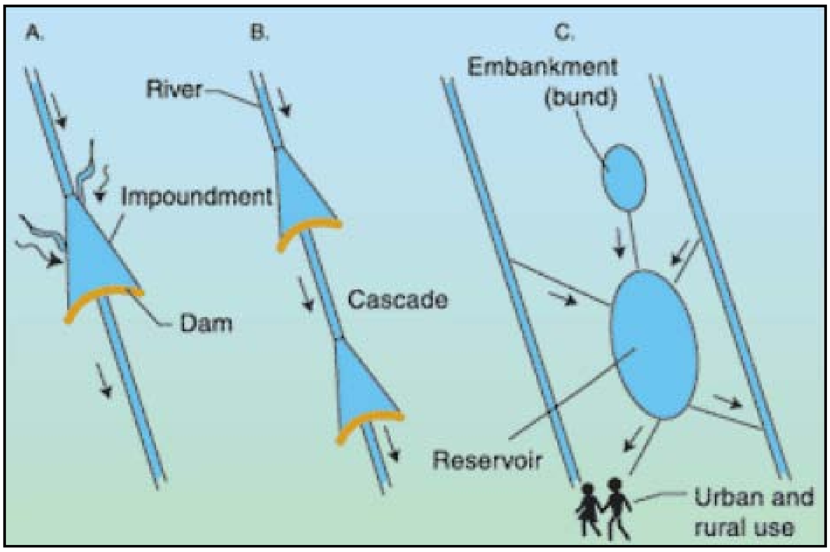

Figure 2.7: Different types of reservoir systems (Rast and Straskraba, 2000).

2.2.2 Morphology of lakes and reservoirs

The shape and size of a lake basin affect nearly all physical, chemical and biological parameters of lakes (Wetzel, 2001). The morphometry of a lake is best described by a bathymetric map (Figure 2.8) which is required for the determination of all major morphometric parameters. Such a map is prepared by a survey of the shoreline by standard surveying methods, often in combination with aerial photography.

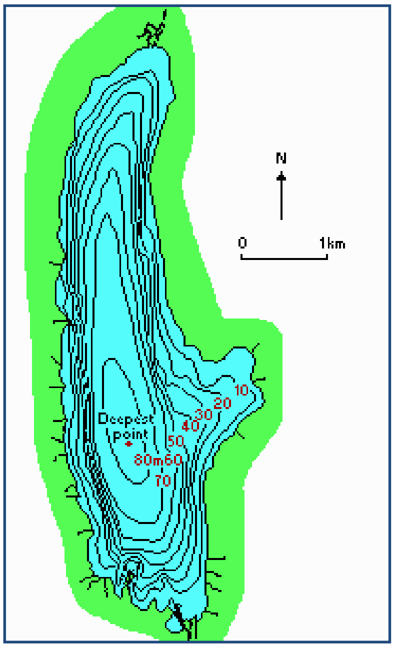

Figure 2.8: Bathymetric map of Lake Ammersee, Germany (ILEC website(a)).

2.2.2.1 Morphometric parameters and curves

Some of the important morphometric parameters include (NALMS et al., 2001; Wetzel, 2001):

- Maximum length, fetch, the distance on the lake surface between the two most distance points on the lake shore, and is considered the maximum effective length for wind to interact on the lake without land interruption.
- Maximum width or breadth, maximum distance on the lake surface at a right angle to the line of maximum length between the shores.
- Surface area.
- Volume.
- Maximum depth.
- Mean depth, lake's volume divided by its surface area.
- Shoreline length (lake circumference).
- Shoreline development, measure of "roundness" of the lake. The ratio of the shoreline length to the circumference of a circle of area equal to that of the lake.
- Hypsographic curve, (depth – area – curve), it is a graphical representation of the relationship of the surface area of a lake to its depth.
- Depth-volume curve, it is a graphical representation of the relationship of the volume of the lake to its depth.
- Hydraulic residence time, retention time, the average time required to completely renew a lake's water volume. May be expressed theoretically as the lake's water volume divided by the rate of total outflow.

Generally, lakes can be classified as shallow (mean depth <7 m) and deep (mean depth >7 m), and can be divided into short residence time, less than one year, and long residence time, more than one year (Ji, 2008).

2.2.2.2 Morphological characteristics of reservoirs

When a new lake is formed by damming a stream, its shape will usually be different from that of most natural lakes. The shoreline development is almost always higher than for natural lakes, because the stream often has a number of tributaries draining into it, when the river is dammed, the impounded water tends to back up into the tributaries (Baxter, 1977). As a result, many reservoirs have a characteristic dendritic shape, with the "arms" radiating outward from the main body of the reservoir. Moreover, natural lakes are normally deepest somewhere near the middle, while river reservoirs are almost always deepest just upstream from the dam, and they have been referred to as *"half-lakes"* (Baxter, 1977). The hydraulic residence time in reservoirs is shorter than the hydraulic residence in lakes.

Reservoirs, formed by the damming of river valleys, possess three distinct zones along the longitudinal gradient (Figure 2.9). Each zone possesses unique and dynamic physical, chemical and biological characteristics (Wetzel, 2001). These zones are:

- ***Riverine zone,*** a narrow basin has high advective water transport and high particulate turbidity.
- ***Transitional zone,*** a broader deeper basin has decreased water velocities as energy dispersed over large areas. Turbidity general decreases and sedimentation increases.
- ***Lacustrine zone,*** the region above the dam has characteristics more similar to natural lakes in relation to stratification and morphological interactions.

2.2.3 Thermal stratification and mixing in lakes and reservoirs

The thermal properties of lakes and the annual circulation events probably have a greater influence on lake biology and chemistry than any other factor (NALMS et al., 2001). Lake water absorbs heat energy directly from sunlight and additional heat from the air. The mixing action of wind helps to distribute this heat throughout a lake's surface water. Because of the strong relationship between water temperature and density (Figure 2.10), density stratification occurs. The maximum water density occurs at approximately 4^oC due to the unique nature of the water molecule. Between 0^oC (freezing) and 4^oC, density nonlinearly increases with increasing temperature. As water temperature increases from 4^oC to 100^oC (boiling), density nonlinearly decreases. A temperature difference of just a few degrees results in a density difference sufficient to prevent most vertical mixing in lakes and reservoirs (Martin et al., 1999).

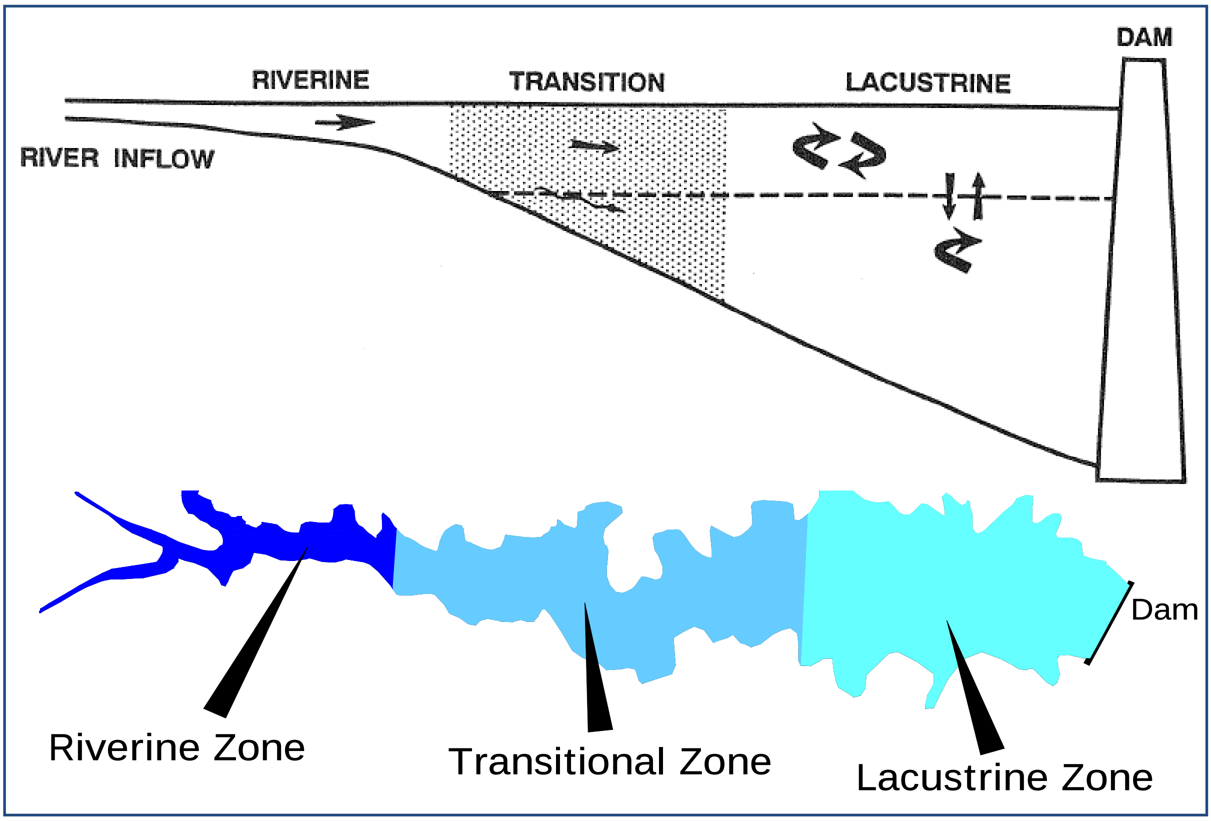

Figure 2.9: Longitudinal zones in reservoirs (Elsammany, 2002; Wetzel, 2001).

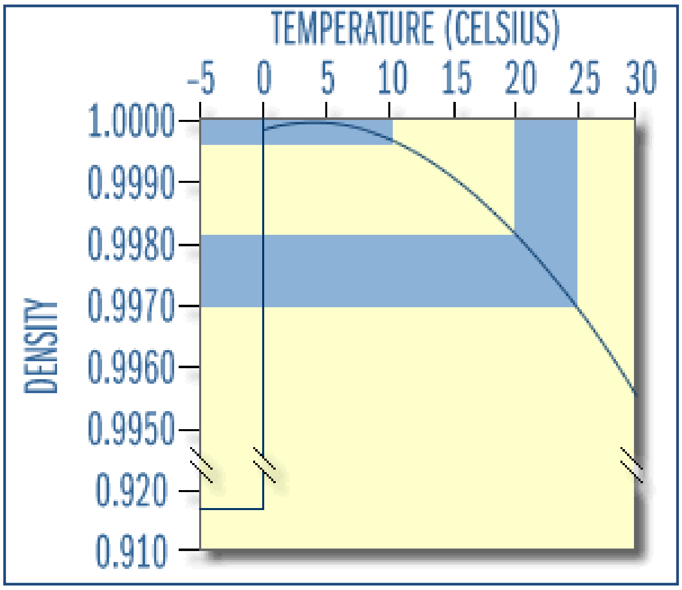

Figure 2.10: Water's temperature-density relationship. The maximum density is at about 4°C. A small change in temperature in warm water results in a larger decrease in density than the same amount of change in cool water (Water On the Web website).

2.2.3.1 Stratification cycle

Figure 2.11 represents the typical seasonal stratification cycle in many lakes and reservoirs. During the winter, if the lake does not freeze, the temperature is relatively constant with depth. As the weather warms in the spring, the top layers of water begin to warm. Since warmer water is less dense, and water is a poor conductor of heat, the lake eventually stratifies into a warm, less dense, surface layer called the *epilimnion* and a cooler, denser, bottom layer, the *hypolimnion*. A thermal gradient, the *metalimnion*, is present between these two layers, the inflection point in the temperature gradient is called the *thermocline* (Weiner and Matthews, 2003). As colder weather approaches, the top layer cools, becomes denser, and sinks. This creates circulation within the lake, known as *fall turnover*. If the lake freezes over in the winter, the lake surface temperature will be less than 4°C, and the ice will float on top of the slightly denser underlying water. When the spring comes, the lake surface will warm slightly and there will be a *spring turnover* as the ice thaws. *Isopleths* (lines of equivalent temperatures – or concentrations) can be used to give an impression of how stratification changes over time (Figure 2.12).

If the lake water has the same temperature from top to bottom, it is said to be isothermal. Under isothermal conditions, wind can mix the entire lake because water from all parts of the lake has the same density (Dodson, 2005).

The phenomenon of thermal stratification depends on several factors, including (Dodson, 2005):

- Time of year, different patterns of annual and daily variation in temperature have a great effect on stratification.
- Lake depth, shallow lakes tend to be easily stirred by the wind, so stratification is more easily destroyed by a wind storm.
- Wind fetch, the longer the fetch, the more likely the wind can mix a lake.
- Topography, adjacent geographic features (such as a bluff or cliff) that shield the lake from wind will contribute to early stratification and to shallow thermocline.
- Solutes, salts (or organic solutes) in lake water make the water denser enough to resist mixing.

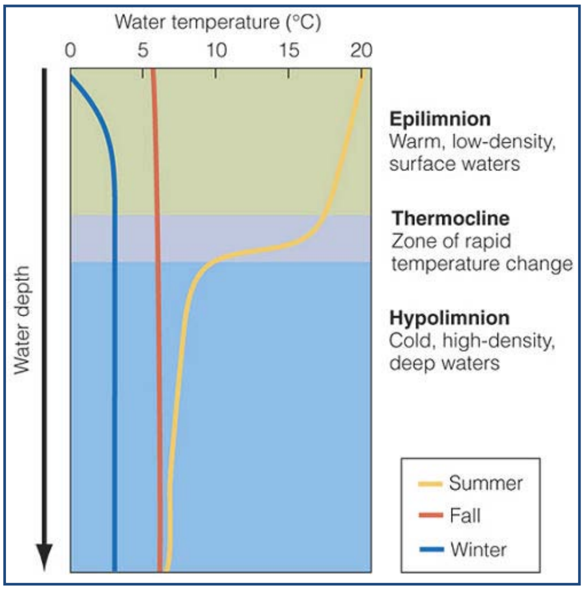

Figure 2.11: Typical temperature–depth relationships in lakes (Smith and Smith, 2006).

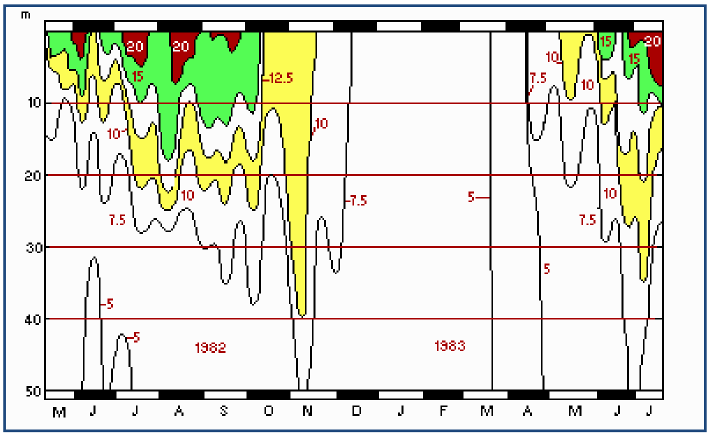

Figure 2.12: Isopleths of water temperature of a warm monomictic lake, Bodensee (Lake Constance), Germany (ILEC (b)).

Martin et al. (1999) indicate that lakes tend to stratify when their mean depths exceed 10 m and mean annual residence times exceed 20 days. Moreover, the modified densiometric Froude number (equation 2.1) provides an indication of the stratification potential of a reservoir,

$$F_d = \sqrt{\frac{1}{ge}\frac{LQ}{D_m V}} \tag{2.1}$$

where:

g	Gravitational acceleration (L T^{-2})
e	Dimensionless density gradient (10^{-6} L^{-1})
L	Reservoir length (L)
Q	Average reservoir discharge (L^3 T^{-1})
D_m	Mean reservoir depth (L)
V	Reservoir volume (L^3)

for $F_d >> 1/\pi$ the reservoir is well mixed, for $F_d << 1/\pi$ the reservoir is expected to be strongly stratified, and for $F_d \approx 1/\pi$ then the reservoir is expected to be weakly or intermittently stratified (Martin et al., 1999).

2.2.3.2 Annual mixing patterns

Thermal characteristics of lakes are a result of climatic conditions that provide a physical classification which is based upon the stratification and mixing characteristics of the water bodies. These characteristics are illustrated in Figure 2.13 and are defined as below.

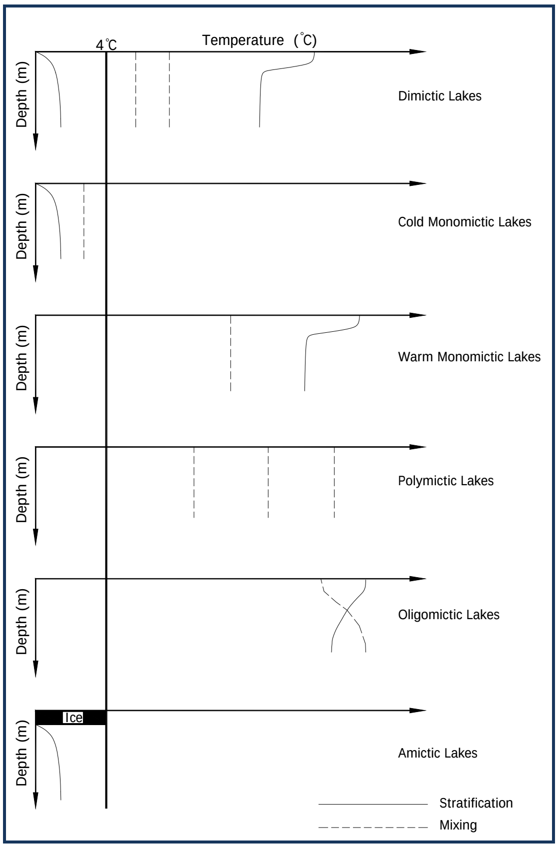

Figure 2.13: Classification of lakes based on thermal stratification and mixing modified after Thomas et al. (1996).

Thermal characteristics of lakes can be classified into (Dodson, 2005; Thomas et al., 1996):

- Dimictic lakes, most lakes in the cooler parts of the temperate zones are dimictic; they stratify twice a year, once in the summer with a warm epilimnion and once in the winter with cold epilimnion relative to the hypolimnion. Mixing occurs twice a year, normally in the spring and autumn.
- Cold monomictic lakes occur in cold areas and at high altitude (sub-polar). The water temperature never exceeds 4°C. They have winter stratification with a cold water epilimnion, often with ice cover for most of the year.
- Warm monomictic lakes, occur in temperate latitudes in subtropical mountains and in areas strongly influenced by oceanic climates. In the same way as their cold water counterparts, they mix only once during the year with temperatures those never fall below 4°C.
- Polymictic lakes, occur in regions of low seasonal temperature variations. All shallow lakes fall within this category. Many tropical lakes are shallow and large polymictic lakes. These lakes may mix each afternoon as the wind rises and then stratify in the morning as the sun warms the upper water.
- Oligomictic lakes, occur in tropical regions and are characterized by rare, or irregular, mixing with water temperatures well above 4°C.
- Amictic lakes, occur in the polar regions and at high altitudes. They are always frozen and never circulate or mix.

- Meromictic lakes, in contrast to holomictic lakes (complete mixing of the entire water column) as described above, meromictic lakes (Figure 2.14) do not undergo complete circulation; the surface water does not mix the lower water layers (Overbeck, 1989). The main reason of this phenomenon is salinity; salinity affects water density much more than does temperature (Dodson, 2005). If enough ions or organic chemicals are dissolved in the hypolimnion, it will be dense enough to resist mixing with the epilimnion. If the deeper layer of water fails to mix, it continues to collect salts or organics and becomes more and more resistant to mixing. The deeper part of the water is termed monimolimnion, underlying the upper part which is termed mixolimnion. Most deep lakes in tropical regions are meromictic, such as Lake Tanganyika and Malawi. Another example of this kind is provided by the Hemmelsdorfersee, Germany, which occupies a crypto depression near the Baltic Sea (Löffler, 2004).

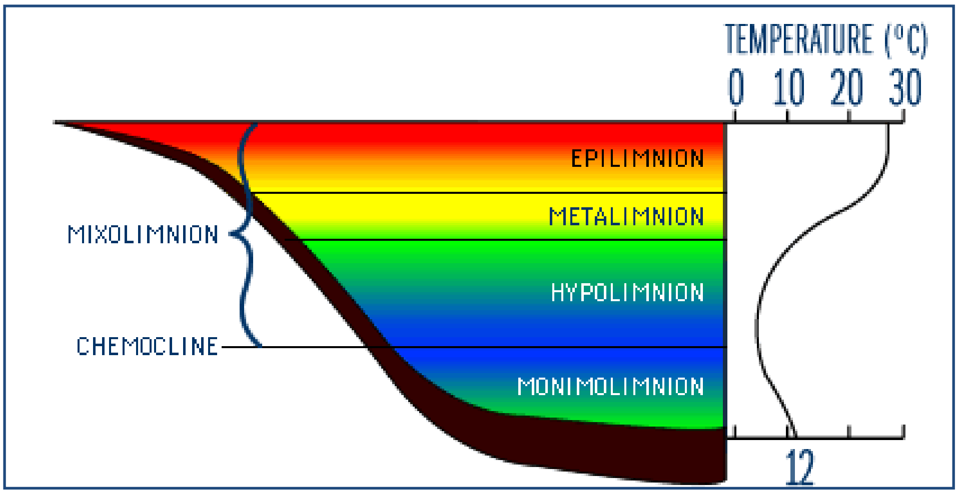

Figure 2.14: Stratification of meromictic lakes (Water On the Web website).

The physical or thermal classification of lakes, as described above, is largely climate controlled and is, therefore, related to latitude and altitude as summarized graphically by Wetzel (2001) (Figure 2.15).

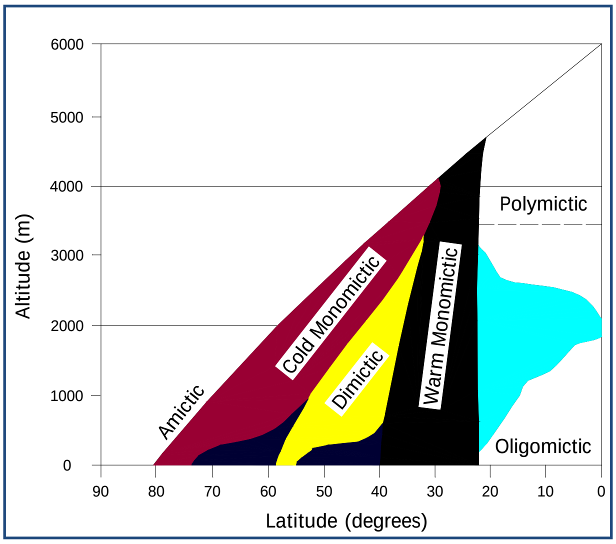

Figure 2.15: The geographic distribution of thermal lake types (Elsammany, 2002).

2.2.3.3 Reservoirs thermal characteristics

Most reservoirs, with long retention times, experience the same thermal structure development as lakes. Many reservoirs are, however, polimictic due to their relatively shallow depths or the effect of enhanced flow induced turbulence. Moreover, some reservoirs are meromictic, this results when a part of the reservoir water that is near the bottom is not mixed with the rest. This is usually related to the construction time, when small coffer dams are built upstream and downstream of the construction site of the major dam. Water becomes trapped and accumulation of chemical constituents takes place (Elsammany, 2002; Straškraba and Tundisi, 1999).

2.2.4 Water quality characteristics

Water quality can be thought of as a measure of the suitability of water for a particular use based on selected physical, chemical and biological characteristics. To determine water quality, scientists first measure and analyze characteristics of the water such as: temperature, dissolved mineral content and number of bacteria. Selected characteristics are then compared to numeric standards and guidelines to decide if the water is suitable for a particular use (Cordy, 2005; USGS, 2001). Even where there is enough water to meet current needs, many rivers, lakes and groundwater resources are becoming increasingly polluted. The most frequent sources of pollution are human waste (with 2 million tons a day disposed of in watercourses all over the world), industrial waste and chemicals and agricultural pesticides and fertilizers (World Water Assessment, 2003). There is an increasing evidence that global climate change and global warming can cause or contribute to degraded water quality (UNEP, 2007).

Some physical, chemical and biological characteristics of lakes and reservoirs water quality are explained here briefly.

2.2.4.1 Physical aspects of water quality

Light

The absorption and attenuation of light by the water column are major factors controlling temperature and potential photosynthesis. Light intensity at the lake surface varies seasonally and with cloud cover and decreases with depth down the water column. The deeper into the water column that light can penetrate, the deeper photosynthesis can occur. The percentage of the surface light absorbed or scattered in a 1 meter long vertical column of water, is called the vertical extinction coefficient. This parameter is symbolized by "*k*". In lakes with low k-values, light penetrates deeper than in those with high k-values (Water On the Web website). Figure 2.16 shows the light attenuation profiles from two lakes with attenuation coefficients of 0.2 m^{-1} and 0.9 m^{-1}.

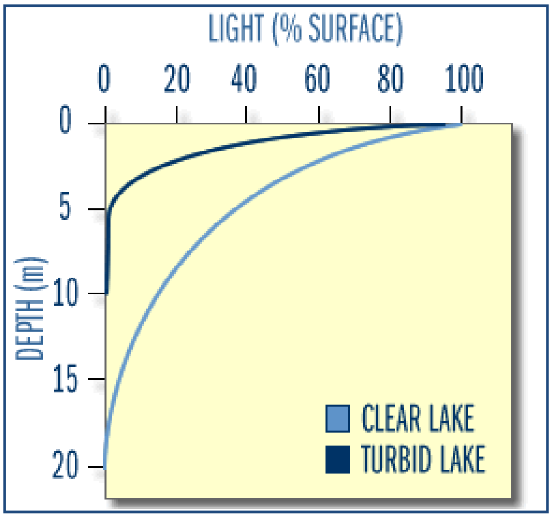

Figure 2.16: Light-depth profiles for a clear lake (k=0.2 m^{-1}) and a turbid lake (k=0.9 m^{-1}) (Water On the Web website).

The maximum depth at which algae and macrophytes can grow is determined by light levels. Limnologists estimate this depth to be the point at which the amount of light available is reduced to 0.5% – 1% of the amount of light available at the lake surface (Water On the Web website). This is called the euphotic zone. As a general rule of thumb; this depth is about 2 to 3 times the limit of visibility as estimated using a Secchi disk.

Temperature

Surface waters temperature is influenced by latitude, altitude, season, time of day, air circulation, cloud cover, the flow and depth of the water body (Chapman and Kimstach, 1996). In turn, temperature affects physical, chemical and biological processes in water bodies and, therefore, the concentration of many characteristics. As water temperature increases, the rate of chemical reactions generally increases together with the evaporation and volatilization of substances from the water. Increased temperature also decreases the solubility of gases in water, such as O_2, CO_2, N_2, CH_4 and others. The metabolic rate of aquatic organisms is also related to temperature, and in warm waters, respiration rates increase leading to increased oxygen consumption and increased decomposition of organic matter. Growth rates also increase leading to increased water turbidity, macrophyte growth and algal blooms, when nutrient conditions are suitable.

Odors

Odors associated with water usually result from the presence of decaying organic matter or, in the case of mineral springs, the reduction of sulfates by bacteria to H_2S gas (Tchobanoglous and Schroeder, 1985). Decaying organic matter may accumulate in bottom deposits large enough to provide suitable conditions for the anaerobic bacteria that produce noxious gases. Sources of the organics include plant debris washed into streams, dead animals, microorganisms and wastewater discharges.

Color

Many of the colors associated with water are not true colors but the result of colloidal suspension. True colors result from dissolved materials, most often organics. Most colors in natural waters result from dissolved tannins extracted from decaying plant material; the result is a slightly brownish tint (Tchobanoglous and Schroeder, 1985).

Total suspended solids (TSS) and total dissolved solids (TDS)

The term "solids" is widely used for the majority of compounds which are present in natural waters and remain in a solid state after evaporation. Total suspended solids *(TSS)* and total dissolved solids *(TDS)* correspond to non-filterable and filterable residue, respectively (Chapman and Kimstach, 1996). High concentrations of total suspended solids *(TSS)* affect light penetration, productivity, turbidity, recreational values and habitat quality. These high concentrations cause lakes to fill in faster. Particles also provide attachment places for other pollutants, notably metals and bacteria (Michaud, 1991).

Turbidity

Perhaps the first characteristic which is noticed about water is its clarity. Very clear natural water allows images to be seen distinctly at considerable depths (Tchobanoglous and Schroeder, 1985). The condition of reduced clarity in water caused by the scattering or absorption of light by suspended particles in coarse or colloidal suspension or the adsorption of light by colored constituents, is known as turbidity (Boyd, 2000). For many years, turbidity was measured by the *Jackson Turbidimeter* or the *Nephelometer* and was reported as Jackson Turbidimeter Unit *(JTU)* or Nephelometer Turbidity Unit *(NTU).* Most natural waters have turbidities less than 50 NTU *(Boyd, 2000; Michaud, 1991)*. Now a relative index of turbidity can be given by using the *Secchi Disk.* A secchi disk is a 20 cm diameter disk painted with alternate black and white quadrants, weighted under the bottom, and attached at its upper face with a calibrated line. The disk is lowered into the water slowly until it just disappears and the depth is measured. The disk is slowly raised until it just appears and the depth is read again. The average of the two readings is the secchi disk visibility. The disk should be read under standard conditions of a clear day, sun behind the observer, between 9 am and 3 pm, and without sunglasses. Secchi disk visibilities range from a few centimeters in highly turbid waters to several meters in clear waters (Boyd, 2000).

2.2.4.2 Chemical aspects of water quality

pH, acidity and total alkalinity

The pH (potential of hydrogen) value is the negative logarithm of the hydrogen ion activity. The pH scale runs from 0 to 14 (i.e., very acidic to very alkaline), with pH 7 representing a neutral condition in water or solution at 25°C. At greater hydrogen ion H^+ concentrations, pH will be below 7.0; the solution is acidic. At greater hydroxide ion OH^- concentrations, pH will be above 7.0; the solution is basic or alkaline. The pH of most natural waters is between 6.0 and 8.5 (Chapman and Kimstach, 1996).

The acidity of water refers to its reserve capacity to generate additional hydrogen ions through various processes. Thus, pH is an intensity factor and acidity is a capacity factor. The relation between pH and acidity is similar to the relationship of temperature to heat content (Boyd, 2000). The acidity of water is controlled by strong mineral acids, strong acids, weak acids and hydrolyzing salts of metals (e.g., iron, aluminum).

The capacity of water to neutralize acid is termed alkalinity. The total alkalinity of water is an index of the total concentration of titratable bases, and it is expressed in milligrams per liter of equivalent calcium carbonate ($CaCO_3$). Total alkalinity concentrations for natural waters may range from 0 mg/L (very low) to more than 500 mg/L (very high) (Boyd, 2000).

Acids and bases can be dangerous to humans and the water environment. Fortunately, acids and bases have the ability to neutralize the effect of each other (Vigil, 2003). Many industries commonly use acids and bases in their production processes. The generated wastewaters from these processes are either acidic or basic and must be neutralized before being discharged in order to prevent water pollution (Vigil, 2003).

Conductivity

The conductivity of a solution is a measure of its ability to conduct an electrical current. Because the electrical current is transported by the ions in solution, the conductivity increases as the concentration of ions increases. Electrical conductance is reported in units of microsiemens per centimeter (ηS cm^{-1}). The conductivity of most freshwaters ranges from 10 to 1,000 ηS cm^{-1}. In polluted waters, conductivity may exceed 1,000 ηS cm^{-1} (Chapman and Kimstach, 1996).

Hardness

The hardness of natural waters depends mainly on the presence of dissolved calcium and magnesium salts. The total content of these salts is known as general hardness, which can be further divided into carbonate hardness (determined by concentrations of calcium and magnesium hydrocarbonates) and non-carbonate hardness (determined by calcium and magnesium salts of strong acids). Hardness may vary over a wide range, calcium hardness is usually prevalent (up to 70%), although in some cases magnesium hardness can reach 50 - 60% (Chapman and Kimstach, 1996).

Major ions (cations and anions)

Major ions (Na^+, K^+, Ca^{+2}, Mg^{+2}, HCO_3^-, Cl^-, SO_4^{-2}) are naturally very variable in surface and groundwater due to local geological, climatic and geographical conditions (Chapman and Kimstach, 1996).

Sodium (Na^+): it is one of the most abundant elements on the earth. Concentrations in surface waters may arise from sewage and industrial effluents and from the use of salts on roads to control snow and ice. Values can range from 1 mg/L, or less to 10^5 mg/L or more in natural brines.

Potassium (K^+): it is found in low concentrations in natural waters since rocks which contain potassium are relatively resistant to weathering. However, potassium salts are widely used in industry and in fertilizers for agriculture and enter freshwaters with industrial discharges and run-off from agricultural land. Concentrations in natural waters are usually less than 10 mg/L, whereas concentrations as high as 100 and 25,000 mg/L can occur in hot springs and brines, respectively.

Calcium (Ca^{+2}): calcium is present in all waters, and is readily dissolved from rocks which are rich in calcium minerals, particularly as carbonates and sulphates, such as limestone and gypsum. Calcium concentrations in natural waters are typically < 15 mg/L.

Magnesium (Mg^{+2}): magnesium is common in natural waters. Natural concentrations of magnesium in fresh-waters may range from 1 to more than 100 mg/L, depending on the rock types within the catchment.

Bicarbonates (HCO_3^-): bicarbonate is the dominant anion in most surface waters. Bicarbonate concentrations in surface waters are usually < 500 mg/L, and commonly < 25 mg/L.

Chloride (Cl^-): it enters surface waters with the atmospheric deposition of oceanic aerosols, with the weathering of some sedimentary rocks (mostly rock salt deposits), from industrial and sewage effluents, and agricultural and road run-off. In pristine freshwaters chloride concentrations are usually lower than 10 mg/L and sometimes less than 2 mg/L. Higher concentrations can occur near sewage and other waste outlets, irrigation drains, salt water intrusions, in arid areas and in wet coastal areas.

Sulphate (SO_4^{-2}): sulphate is naturally present in surface waters. Industrial discharges and atmospheric precipitation can also add significant amounts of sulphate to surface waters. Sulphate can be used as an oxygen source by bacteria which convert it to hydrogen sulphide (H2S and HS^-) under anaerobic conditions. Sulphate concentrations in natural waters are usually between 2 and 80 mg/L, however, they may exceed 1,000 mg/L near industrial discharges or in arid regions where sulphate minerals, such as gypsum, are present. High concentrations (> 400 mg/L) may make the water unpleasant to drink.

Dissolved oxygen (DO)

Oxygen is essential to all forms of aquatic life. The oxygen content of natural waters varies with temperature, salinity, turbulence, the photosynthetic activity of algae and plants and atmospheric pressure. The solubility of oxygen decreases as temperature and salinity increase (Figure 2.17). In fresh-waters, dissolved oxygen (DO) at sea level ranges from 15 mg/L at 0°C to 8 mg/L at 25°C. Concentrations in unpolluted waters are usually close to, but less than, 10 mg/L (Chapman and Kimstach, 1996).

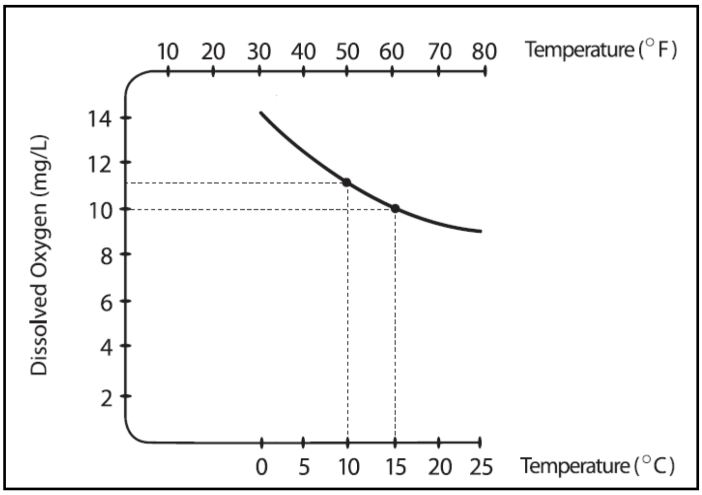

Figure 2.17: DO-Temperature (Vigil, 2003).

Variations in DO can occur seasonally, or even over 24 hour periods, in relation to temperature and biological activity (i.e. photosynthesis and respiration) (Figure 2.18). Biological respiration, including that one related to decomposition processes, reduces DO concentrations. In severe cases of reduced oxygen concentrations (whether natural or man-made), anaerobic conditions can occur (i.e. 0 mg/L of oxygen), particularly close to the sediment-water interface as a result of decaying material. Concentrations below 5 mg/L may adversely affect the functioning and survival of biological communities and below 2 mg/L may lead to the death of most fish (Chapman and Kimstach, 1996).

Patterns of vertical oxygen profile,

Oxygen concentrations depend on thermal stratification and biological activity. There are three major patterns of concentration variation with depth (Dodson, 2005):

- An orthograde pattern results if a lake is saturated with oxygen at all depths and has a warm epilimnion. It will have a higher oxygen concentration in the hypolimnion than in the epilimnion (Figure 2.18).
- A clinograde pattern results from bacterial decomposition and the associated oxidation of organic material in the hypolimnion. In the summer, a stratified, productive lake will typically have oxygen concentrations near or exceeding saturation in the epilimnion, and little or no oxygen in the hypolimnion (Figure 2.18). This is because phytoplankton supply oxygen to the epilimnion, which is mixed, while decomposition removes oxygen from the unmixed hypolimnion. As productivity increases due to algal growth, lake water becomes more turbid, light penetrates less deeply and more organic material falls into the hypolimnion. Bacterial metabolism of this organic material during the summer can reduce hypolimnetic oxygen concentrations to zero. In meromictic lakes, extreme clinograde oxygen profiles occur.
- A heterograde pattern is a profile with a peak (maximum oxygen concentration) at an intermediate depth (Figure 2.19). This pattern is characteristically for low-productivity lakes, in which the algae community is so rarefied that light penetrates into the metalimnion or

hypolimnion. Algae in the unmixed hypolimnion flourish, receiving light from above, and nutrients from the sediments. The algae often form a visible but thin and insubstantial layer (Figure 2.20) that is about 1 meter thick, deep in the lake.

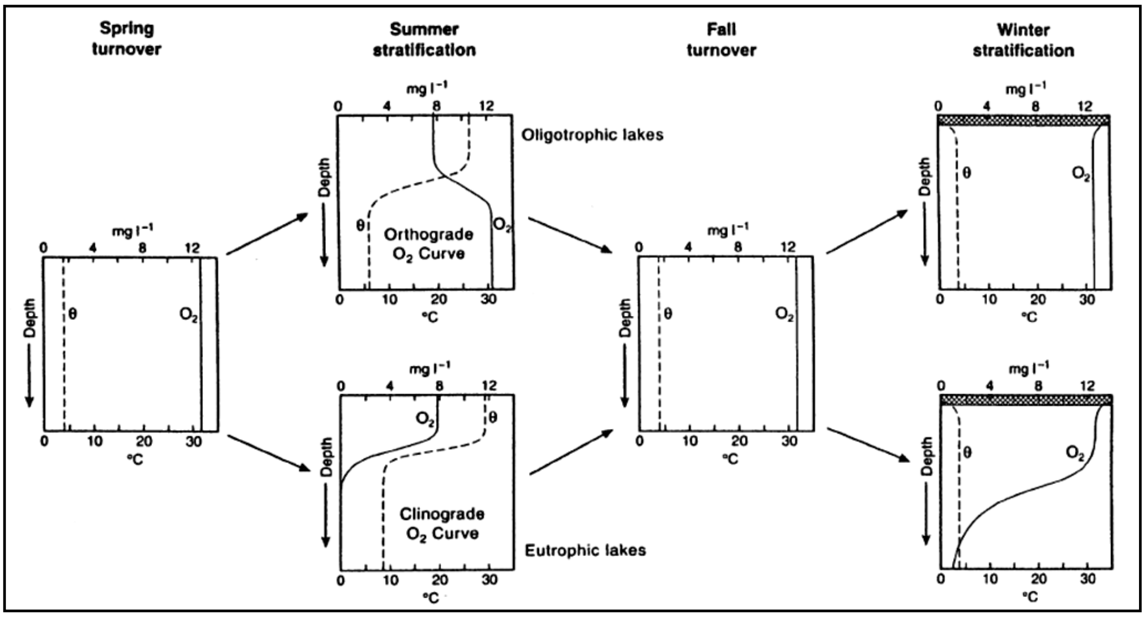

Figure 2.18: Idealized vertical distributions of oxygen concentrations and temperature during the four main seasonal phases of an oligotrophic and eutrophic dimictic lake (Wetzel, 2001).

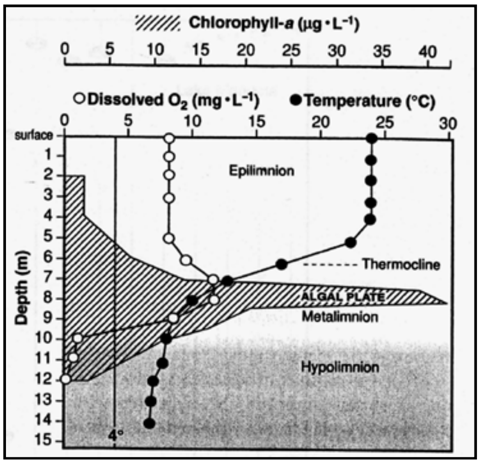

Figure 2.19: Vertical profiles of chlorophyll a, oxygen concentration and temperature for oligotrophic Jack's Lake, Ontario, Canada (Dodson, 2005).

Figure 2.20: Vertical phytoplankton samples from a lake in Ontario, July 1985. The samples demonstrate a hypolimnetic algal plate. The first (to left) clear sample is from the hypolimnion, below the algal plate. The dark-green sample is from the algal plate, and the light-green sample was taken just 1 meter above. The next three clear samples are from the epilimnion (Dodson, 2005).

Organic Matter

Most freshwaters contain organic matter (materials are made from carbon, such as plants and animals), which can be measured as total organic carbon (TOC). The total organic matter in the water can be a useful indication of the degree of pollution. The impact of pollutants on the oxygen resources of a stream or lake is a major factor in the development of any water quality management plan. The impact is normally measured as *oxygen demand,* a parameter that can be interpreted as a gross measure of the concentration of oxidizable materials present in a water sample. The most two tests which are used most often are the measure of biochemical oxygen demand (BOD) (is used for municipal wastewaters) and chemical oxygen demand (COD) (is used for industrial wastewaters) (Vigil, 2003).

In surface unpolluted waters, total organic carbon (TOC) concentrations are generally less than 10 mg/L, BOD values of 2 mg/L O_2 or less, and COD concentrations range from 20 mg/L O_2 or less. In polluted waters, TOC concentrations may exceed 100 mg/L, BOD may exceed 10 mg/L O_2 and COD may be greater than 200 mg/L O_2 (Chapman and Kimstach, 1996).

Nutrients

Nutrients are the elements that all living organisms need for growth. They are the nutritional building blocks of bacteria, fish, trees and humans. Nutrients come in both organic and inorganic forms. The most important nutrients in water quality and water pollution control are carbon, nitrogen and phosphorus (Vigil, 2003):

Carbon,

Carbon is one of the most abundant elements on Earth. It exists in a number of chemical forms in water (Figure 2.21), including (Dodson, 2005):

Carbon dioxide (CO_2): it is a small component of the Earth's atmosphere dissolves in water. In aquatic systems, CO_2 is consumed by photosynthesis and produced by metabolism (respiration).

Dissolved inorganic carbon (DIC): it includes carbon dioxide, carbonic acid, bicarbonate and carbonate.

Dissolved organic carbon (DOC): methane (CH_4), the simplest form of organic carbon, is produced by anaerobic decomposition.

Particulate organic carbon (POC): any discarded parts of all living and dead organisms (such as dead leaves).

Inorganic carbon is a major nutrient of photosynthetic metabolism. However, phosphorus and nitrogen limit photosynthesis more frequently than does inorganic carbon, which occurs in much greater abundance.

When too much carbon is released into an aquatic system in an uncontrolled manner, such as discharging untreated municipal sewage or industrial waste, it causes water pollution. One of the ways which can prevent substances containing carbon from polluting our waters is by constructing municipal and industrial wastewater treatment plants (Vigil, 2003). These treatment plants remove most of the materials containing carbon from the wastewater before it is discharged.

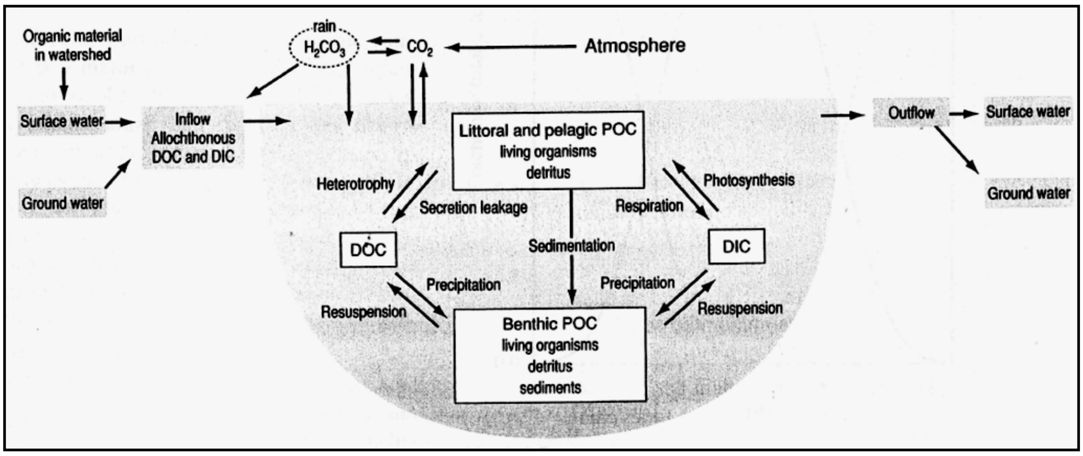

Figure 2.21: Typical carbon cycle in lakes (Dodson, 2005).

Nitrogen,

Nitrogen is also an abundant element in the environment and it is an essential nutrient for all living organisms. Plants and micro-organisms convert inorganic nitrogen to organic forms. Nitrogen occurs in several different chemical forms in water, including:

Total Nitrogen (TN): it includes all inorganic and organic forms of nitrogen, but doesn't include nitrogen gas dissolved in the lake water. Typical concentrations of TN in lakes range from 0.4 to 2.7 µg/l (Wetzel, 2001).

Nitrogen gas (N_2): a relatively inert gas that makes up most of Earth's atmosphere (about 78% of the atmosphere) and dissolves readily in water.

Dissolved inorganic nitrogen (DIN): it is the sum of nitrate, nitrite and ammonium dissolved in the water.

Nitrate and nitrite (NO_3^-) and (NO_2^-): the nitrate ion is the common form of combined nitrogen found in natural waters. It may be biochemically reduced to nitrite by denitrification processes, usually under anaerobic conditions. Nitrate is used by many organisms, including plants, as a source of nitrogen for building proteins and other organic nitrogen compounds. Natural sources of nitrate to surface waters include igneous rocks, land drainage and plant and animal debris. Nitrite, on the other hand, can be quite toxic. Determination of nitrate plus nitrite in surface waters gives a general indication of the nutrient status and level of organic pollution. Nitrate concentrations in surface water are, less than 0.1 mg/L NO_3-N for natural water, less than 1 mg/L NO_3-N to 5 mg/L NO_3-N for unpolluted, human used water and more than 5 mg/L NO_3-N for polluted water by human or animal waste, or fertilizer run-off. Nitrite concentrations in freshwaters are usually very low, 0.001 mg/L NO_2-N, and rarely higher than 1 mg/L NO_2-N (Chapman and Kimstach, 1996).

Ammonia (NH_3): it is the reduced form of nitrogen that is made by nitrogen fixation, either bacteria or in an industrial process. Ammonia is also a waste product, used by many organisms to get rid of excess nitrogen or hydrogen ions, or both. It dissolves in water, combines with hydrogen ions to produce the ammonium ion (NH_4^+). When oxygen is present, ammonia spontaneously oxidizes to nitrate. At certain pH levels, high concentrations of ammonia are toxic to aquatic life. Unpolluted waters contain small amounts of ammonia and ammonia compounds, usually less than 0.2 mg/L N (Chapman and Kimstach, 1996).

Organic nitrogen compounds consist mainly of protein substances (e.g., amino acids, nucleic acids and urine) and the product of their biochemical transformations (e.g., humic acids and fulvic acids), formed in water by phytoplankton and bacter*ia.*

Six major biological processes involving nitrogen in fresh waters are (Figure 2.22):

Nitrogen fixation: N_2 gas and chemical energy are converted to ammonium, by blue-green algae or bacteria.

Nitrification: is the conversion of reduced forms such as ammonium to nitrite or nitrate, under aerobic conditions.

Dinitrification: nitrate (partial reduction) is converted to nitrogen gas, by bacteria under anoxic conditions.

Assimilation: DIN or organic nitrogen is incorporated into organic compounds, by plants and micro-organisms.

Excretion: animals typically ingest too much nitrogen (as amino groups) and need to excrete the nitrogen as ammonium ion, urea, or uric acid.

Remineralization: the conversion of particulate organic matter (POM) (such as organic detritus) to dissolved organic nitrogen (DON) by heterotrophic bacteria.

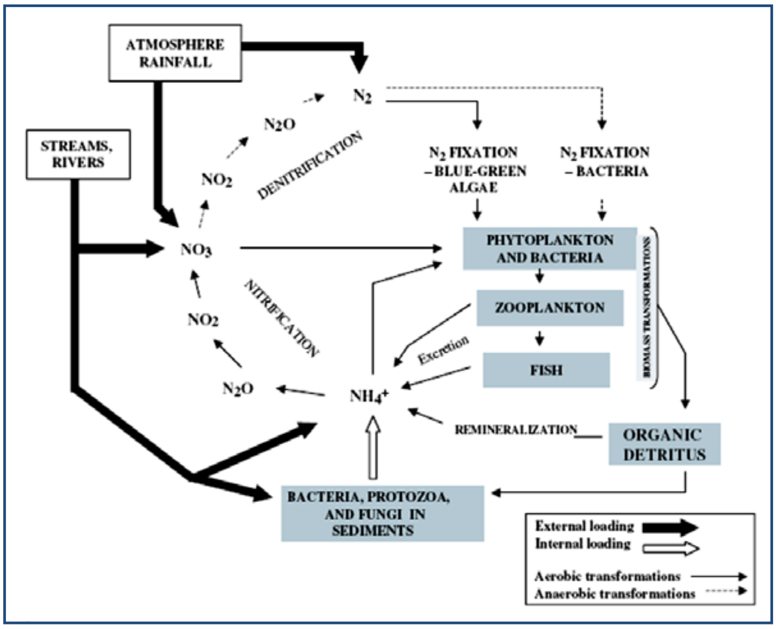

Figure 2.22: Typical nitrogen cycle in lakes (Sigee, 2005).

Phosphorus,

Phosphorus is an essential nutrient for living organisms. It is generally the limiting nutrient for algal growth and, therefore, controls the primary productivity of a water body. Natural sources of phosphorus are the weathering of phosphorus-bearing rocks and the decomposition of organic matter. Domestic waste-waters (particularly those containing detergents), industrial effluents and fertilizer run-off contribute to elevated levels in surface waters. Phosphorus associated with organic and mineral constituents of sediments in water bodies can also be mobilized by bacteria and released to the water column. Phosphorus is present in several major forms in lakes (Figure 2.23):

Total phosphorus (TP): includes all the phosphorus in a sample of water and is determined by converting all the phosphorus (soluble or particulate) in an unfiltered sample into inorganic orthophosphate. TP is the preferred indicator of a lake's nutrient status. Typical concentrations of TP in lakes range from 10 µg/L to 80 µg/L (Wetzel, 2001), while in polluted water may reach 200 µg/L (Dodson, 2005).

Soluble reactive phosphorus (SRP): includes forms of phosphorus (inorganic and organic) that are dissolved in water and are readily available for uptake by algae and macrophytes.

Total soluble phosphorus (TSP): includes the phosphorus (organic or inorganic) that passes through a filter. TSP includes SRP and any soluble, nonreactive phosphorus.

Orthophosphate – or phosphate ion (PO_4^{3-}): is the inorganic form of phosphate, and it is the major component of SRP. Dissolved orthophosphate is also called dissolved inorganic phosphorus (DIP).

Dissolved organic phosphorus (DOP): is produced by living or decomposing organisms.

Particulate organic phosphorus (POP): includes phosphorus in organic chemicals in living or dead organisms, such as dead leaves.

Particulate inorganic phosphorus (PIP): orthophosphate attached (adsorbed) to particles.

Phosphine gas (PH_3): the poisonous phosphine gas can be produced by anaerobic bacteria decomposition of organic matter (in anoxic condition).

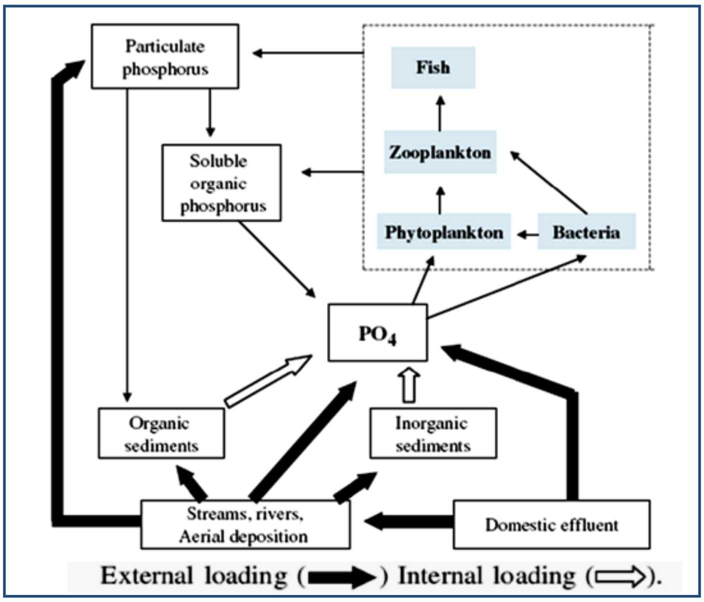

Figure 2.23: Typical phosphorus cycle in lakes (Sigee, 2005).

Trace Elements

In addition to macronutrients (inorganic compounds containing carbon, nitrogen, phosphorus and silicon) natural waters also contain a variety of dissolved chemicals containing elements which are essential but are required in smaller quantities. These micronutrients include some of the anions and cations, plus a range of (trace) elements occurring at much smaller concentrations (Sigee, 2005).

Trace elements include the metals manganese, iron, cobalt, copper, zinc, selenium, molybdenum and barium, plus non-metals such as fluorine. Other elements, such as vanadium and cadmium are also frequently included on the list, but their primary functions and absolute requirements are not clear. The concentrations of trace elements in natural waters vary with water chemistry and temporal (seasonal and diurnal) factors (Table 2.1).

Table 2.1: Range in average concentrations (µg/L) of soluble trace elements in surface waters of lakes and rivers (Wetzel, 2001).

Element	Fe	Mn	Cu	Zn	Co	Mo	V
Range	1 - 540	0.3 – 140	1 - 10	1 - 13	0.05 - 0.9	0.4 - 30	0.03 - 0.1

Table 2.2 shows the average concentrations of trace elements in the epilimnion, hypolimnion, phytoplankton and sediments of Schöhsee, Northern Germany (Sigee, 2005).

Table 2.2: Average concentrations of trace elements in the epilimnion, hypolimnion, phytoplankton and sediments of Schöhsee, Northern Germany (Sigee, 2005).

	Mn	Fe	Cu	Zn	Co	Mo
Epilimnion (E) $\mu g\ l^{-1}$	4.5	15	1.0	1.8	0.03	0.21
Hypolimnion (H) $\mu g\ l^{-1}$	590	425	0.9	1.9	0.07	0.30
Enrichment ratio H/E	130	28	0.9	1.1	2.3	1.4
Phytoplankton (P) $\mu g\ g^{-1}$	130	950	60	110	1.1	4.2
Enrichment ratio P/E	29×10^3	63×10^3	60×10^3	61×10^3	37×10^3	20×10^3
Sediments (S) $\mu g\ g^{-1}$	1600	58 000	95	350	8.3	1.4
Enrichment ratio S/E	355×10^3	3900×10^3	95×10^3	195×10^3	280×10^3	6.7×10^3

2.2.4.3 Biological aspects of water quality

Biological zones in lakes and reservoirs,

Based on its biological communities and linked to its physical structure, a lake can be separated into three distinct biological zones (Figure 2.24)*: littoral* zone, *limnetic (or pelagic)* zone and *benthic* zone (Ji, 2008).

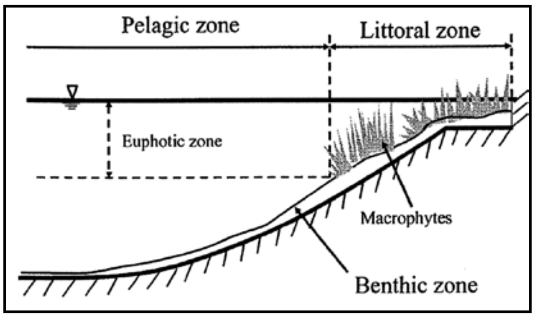

Figure 2.24: Typical biological zones in lakes (Ji, 2008).

The littoral zone is the near shore area where sunlight penetrates all the way to the sediment and allows rooted and floating aquatic plants (macrophytes) to grow. Light levels of about 1% or less of surface values usually define this depth. The 1% light level also defines the euphotic zone of the lake, which is the layer from the surface down to the depth where light levels become too low for photosynthesizers. In most lakes, the sunlight euphotic zone occurs within the epilimnion. The depth of the euphotic zone is about two to three times that of the Secchi depth (Ji, 2008).

The limnetic (or pelagic) zone is the open water area where light does not generally penetrate all the way to the bottom. The bottom sediment, known as the benthic zone, it is the thin sediment layer on the lake bottom. It is typically a few centimeters (about 2 - 5 cm) thick and contains a wide variety of benthic organisms.

Primary biological processes in lakes and reservoirs,

Three primary biological processes occur in lakes, *photosynthesis, respiration* and the *bacterial decomposition* of organic material (Figure 2.25).

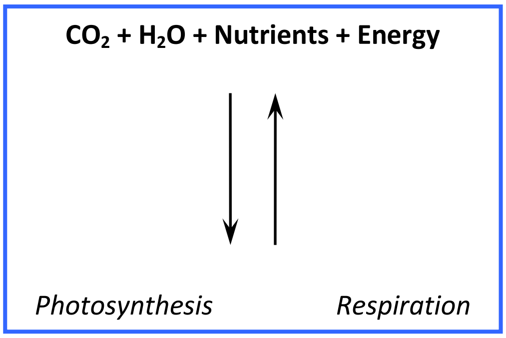

Figure 2.25: An equilibrium relationship between photosynthesis respiration-decomposition in lakes, modified after NALMS et al. (2001).

Photosynthesis involves the conversion of nonliving inorganic chemicals, with energy from sunlight, into living organic plant tissue (organic matter). Lake photosynthesizers include algae and macrophytes, together they are the primary producers, because they create the organic material required by most other organisms for nutrients and energy (Water On the Web website). The amount of plant material produced and remaining in the system is called primary production and the rate of photosynthetic uptake of carbon is called primary productive. The green pigment chlorophyll (which exists in three forms: chlorophyll *a, b and c*) is present in most photosynthetic organisms. It provides an indirect measure of algal biomass and an indication of the trophic status of a water body.

Aerobic respiration is the reverse of photosynthesis. Plants, animals and some bacteria use oxygen to oxidize organic matter and gain energy from the process (Warburton, 2003).

In most natural waters, a slight excess of photosynthesis over respiration exists. This creates excess dead organic matter that is not oxidized. In lakes this organic matter settles to the bottom. If enough organic matter is present, the water becomes murky, preventing light from reaching deep into the lake. In deep or murky lakes photosynthesis does not occur where light does not penetrate. Respiration still occurs and may consume most or all available oxygen. If the oxygen is exhausted before the organic matter, the excess organic matter in the hypolimnion will be buried in the bottom sediments. When the hypolimnion has no dissolved oxygen, anaerobic bacteria begin to play a major role. Reduction of oxidized colloidal particles takes place (organic carbon is partially converted to methane, nitrate ion may be reduced to nitrogen, and so on). These anoxic reactions lead to the great stench in some lakes on turnover (Warburton, 2003).

Living organisms in lakes and reservoirs,

The distribution and abundance of living organisms in lakes and reservoirs depend, partially, on the vertical profiles of temperature, oxygen and light. The main living organisms in lakes and reservoirs are (Dodson, 2005; Ji, 2008; NALMS et al., 2001):

- *Bacteria:* are a diverse group of microscopic organisms. They may live free in the water or they may be attached to surfaces or live in the bottom sediment. The food source for most bacteria (heterotrophic) is organic matter obtained from the environment, but a few bacteria are capable of photosynthesis (autotrophic). The major role of bacteria is to decompose organic matter to its essential inorganic components.
- *Algae*: are photosynthetic organisms that form the base of the aquatic food chain and are grazed upon by zooplankton and herbivorous fish. Microscopic algae that have little or no resistance to water currents and live suspended in the water are called phytoplankton. Other lake algae which may be found primarily attached to substrates in the water such as rocks and rooted plants are known as periphyton. The abundance (production) of photosynthetic algae is controlled primarily by water temperature, light, nutrients, hydraulic residence time and consumption by herbivorous predators. The abundance of phytoplankton can be determined by measuring the amount of chlorophyll pigments in lake water. When there is enough light for photosynthesis, the availability of nutrients often controls phytoplankton productivity. Phosphorus and nitrogen are the most used elements, therefore, they are the ones which affect lake productivity. Algae are often grouped based algal adaptability to environmental conditions, such as temperature, light and nutrient conditions. Groups of algae include blue-green algae (cyanobacteria), green algae and diatoms. Figure 2.26 shows their typical seasonal variations. Except for their chlorophyll-based photosynthesis, blue-green algae are actually bacteria. They have several characteristics that enable them to dominate and create nuisance or noxious conditions. Some blue-green algae are capable of fixing nitrogen gas (N_2), which is derived from the atmosphere and dissolved in the water, as a nitrogen source. This fixation allows them to grow when other algal forms are starved for nitrogen.
- *Macrophytes*: are Macro-algae and flowering water plants attached to the bottom. Macrophytes obtain their nutrients from the bottom sediments rather than the water and are restricted by light to the shallow littoral water.
- *Zooplankton*: are small (microscopic and macroscopic) animals that live in open water (the pelagic or planktonic zone). They are the primary consumers of algae; they transmit the energy to organisms at higher trophic levels in the food chain.

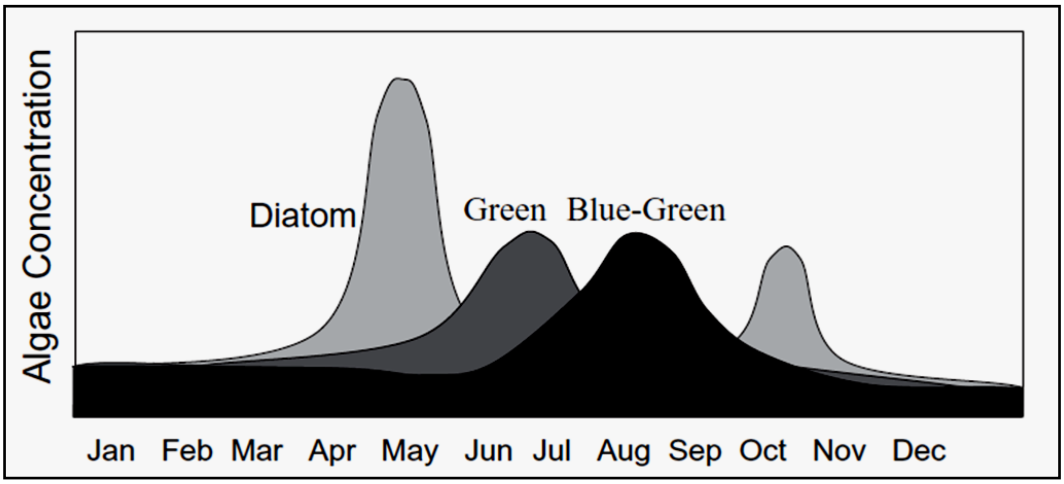

Figure 2.26: Typical seasonal variations of algal concentration in lakes (Ji, 2008).

2.2.5 Water quality issues

A water quality issue may be defined as a water quality problem or impairment which inhibits or prevents some beneficial water use (Thomas et al., 1996). Problem of lakes and reservoirs are caused by the human use of these ecosystems. Most lakes have multiple uses and the problems are related to the conflict between these uses. Jørgensen and Vollenweider (1989) summarized the sources or causes of lake problems as follows:

- Discharge of organic biodegradable wastes.
- Discharge of nutrients from waste water.
- Non-point pollution of nutrients, mainly coming from agriculture.
- Acid rain (caused by air pollutants: SO_2 and NO_x).
- Discharge of toxic substances from industries or agriculture.
- Thermal discharges.

2.2.5.1 Eutrophication

Eutrophication (literally means well-fed or over-fed) refers to the process of high primary biological production caused by excessive availability of nutrients in a lake or reservoir, and all its consequences (Stefan, 1994). Eutrophication could be *natural* or *cultural* (due to human activities). Natural eutrophication takes thousands of years to occur, whereas cultural eutrophication takes only a few decades (or even years) to develop. Eutrophication results from accelerated nutrient loading from expanded industrial activity, added wastewater discharges as in the case of Lake Washington (USA) and Lake Vänern (Sweden), or expanded farming practices combined with sewage discharge as identified for Lake Titicaca, Peru and Bolivia (Dinar et al., 1995). When eutrophication occurs, the waterbody becomes overwhelmed by excessive nutrients, such as nitrogen and/or phosphorus. The excessive nutrients produce more phytoplankton than can be consumed by the waterbody. This overproduction can lead to a variety of problems (Figure 2.27) including (Ji, 2008):

- Low DO, especially near the bottom of the water body.

- High suspended solids often enriched with organic material.
- High nutrient concentrations.
- High algal concentrations.
- Low light penetration and low water clarity.
- Odors from algae or anaerobic mud.
- Changes in species composition.

Recovery of eutrophication is very slow. In some lakes, it has taken up to ten years after external sources of phosphorus were terminated for full recovery of the trophic category of the lake (e.g., from eutrophic to Mesotrophic) (Straškraba and Tundisi, 1999).

When nutrients (mainly; phosphorus and nitrogen) are introduced into the lake, either naturally from storm runoff, or from a pollution source, rapid growth of algae in the epilimnion occurs. When the algae die, they drop to the lake bottom (the hypolimnion) and become a source of carbon for decomposing bacteria. As more and more algae die, all the available dissolved oxygen in the hypolimnion (and also in the metalimnion) will be used by the aerobic bacteria in the decomposition process. When this occurs, anaerobic bacteria activity will start in the hypolimnion (Weiner and Matthews, 2003).

Lakes and reservoirs, according to their biological productivity and nutrient conditions, are commonly grouped into three different trophic states (Figure 2.28):

- Oligotrophic lakes (low nutrients/low productivity).
- Mesotrophic lakes (intermediate nutrients/intermediate productivity).
- Eutrophic lakes (high nutrients/high productivity).

The growth of algae in most lakes is limited by the availability of phosphorus; if phosphorus is in sufficient supply, nitrogen is usually the next limiting nutrient. A limiting nutrient is an essential element or compound that controls the rate of algal growth because the nutrient is not readily available. In waters with N/P ratio greater than 7 to 10, phosphorus will be limiting, whereas nitrogen will be limiting in lakes with a N/P ratio below 7 (Thomas et al., 1996). In the case of nitrogen is the limiting nutrient, cyanobacteria, or blue green algae, store excess phosphorus inside their cells in a process called *luxury consumption* to support future cell growth. Moreover, cyanobacteria have the ability to use dissolved nitrogen gas as a nitrogen source, where most of the other algae cannot use it. As a result, cyanobacteria thrive in environment where nitrogen has become limiting to other algae. Cyanobacteria are often water quality indicators of phosphorus pollution (Weiner and Matthews, 2003).

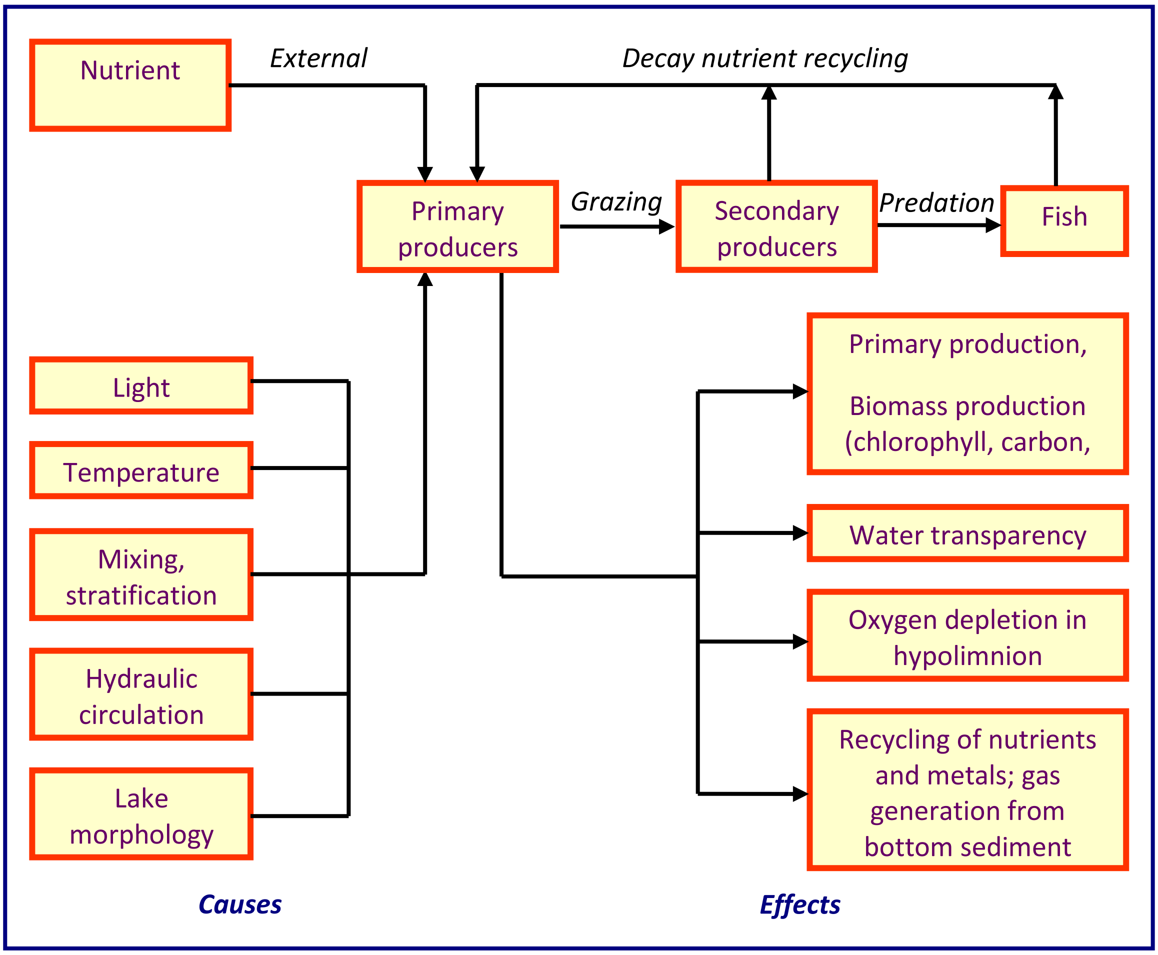

Figure 2.27: Causes and effects of eutrophication, modified after Thomas et al. (1996).

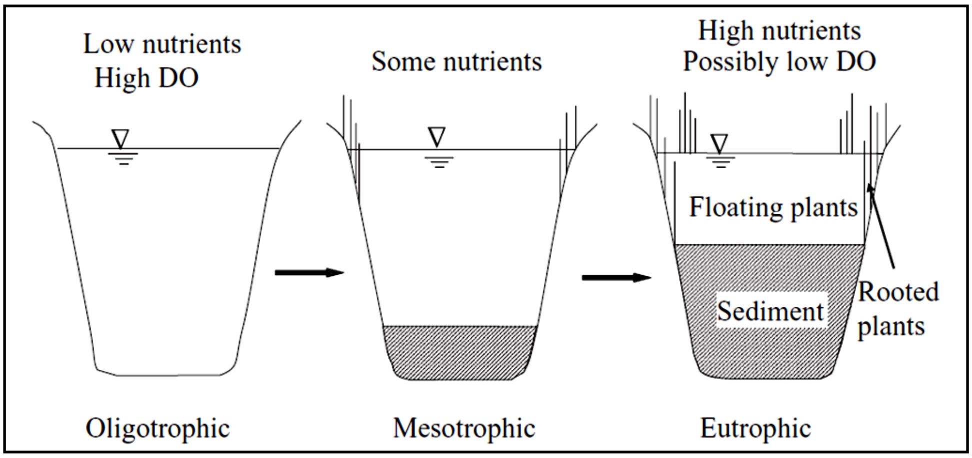

Figure 2.28: The trophic status of lakes and reservoirs (Ji, 2008).

2.2.5.2 Acidification

Acidification is defined as a decrease in pH. The main cause of it is acid rain which is created due to the atmospheric pollution by sulfur dioxide and nitrous oxides. Other sources of acid deposits are industrial effluents and mining wastes. Associated to acidification are changes in leaching of heavy metals (particularly the toxic form of aluminum) from soil into ground- and surface-waters. By 1986, 70% of Norwegian lakes have been acidified by acid rain from central and western Europe with pH below 4.7 (Warburton, 2003). In southern Norway, acidification has caused more than a third of the lakes to become fishless (Dinar et al., 1995).

To neutralize the acidity of lakes, lime can be applied. Limestone dissolved slowly, resulting in a gradual increase in pH value. However, liming is expensive and must be repeated. In Sweden liming has been used extensively over a long period. It has shown reasonably good results (Dinar et al., 1995).

2.2.5.3 Salinisation

Salinisation is becoming a widespread water quality issue which often leads to the loss of usable water resources (Thomas et al., 1996). Natural salinisation occurs over long periods of time and is caused by a change in the natural water balance of the lake. In northeast Brazil, annual evaporation of approximately 2000 mm exceeds the average maximum precipitation of 1200 mm. Therefore, rapid increase in salinity occurs within reservoirs and results in undesirable consequences on multiple uses (Straškraba and Tundisi, 1999). Another causes of salinisation include, excessive fertilizer application on land and soil irrigation in arid and semi-arid regions. Also the input of saline groundwater, results in a meromictic condition for normal lakes. In shallow lakes, however, mixing may occur leading to have lakes with brackish water or moderate salt composition. This condition often exists in coastal lakes, particularly when over-use of fresh ground water allows marine salt water intrusion (Thomas et al., 1996). Saline lakes are often alkaline, the pH may exceed 10 (Warburton, 2003).

2.2.5.4 Sedimentation

Sedimentation (or siltation) of lakes and reservoirs occurs normally and very slowly. However, activities in a lake's watershed that disturb the soil can accelerate this process. For example, Davy Crockett Lake behind Nolichucky Dam in Tennessee was completely filled with sediment from improper mining activities in its watershed within a few decades of building the dam. This lake was designed to provide beneficial uses for more than 100 years. The hydropower-generating capacity of the dam has been totally lost. The lake, no longer serves as a water source for nearby communities. Fishing and other recreational activities have been vastly diminished (Dinar et al., 1995). To correct this problem, dredging is one of the most frequently techniques, however, it is costly. *Dinar et al. (1995)* showed that dredging has been successfully used in Lake Trummen (Sweden) and in Lake Vancouver (USA).

2.3 Water quality modeling

Modeling can be defined as the process of application of fundamental knowledge or experience to simulate or describe the performance of a real system to achieve certain goals (Nirmalakhandan, 2002), and a model is any analysis tool that reduces a physical system to a set of equations or a reduced-scale physical model (Socolofsky and Jirka, 2002).

2.3.1 Modeling objectives and applications

It is important to remember an immutable fact about modeling: *"All models are wrong, but some are useful"!* Models can be useful because they enable the likely effects of management actions and climate scenarios to be simulated, so comparisons can be made between different options "virtually" (Toolkit). Models and simulations also allow rapid and varied evaluation of causes and effects and the principal advantage is that they enable an analysis of even long-term actions with limited investment costs (Makinia et al., 1998).

Generally, the objectives of modeling can be classified to:

- Research-oriented, (simulation) to develop an understanding of the processes and interactions affecting water quality.
- Management-oriented, (optimization) one or more of the following: to manage, operate, or control the system to achieve desired outcomes; to design methods to improve or modify the system; to test hypotheses about the system; or to forecast its response under varying conditions.

Water quality models are used extensively for the following applications (Palmer, 2001):

- In the approval process for a new discharge outfall or intake, for changes to a wastewater treatment system, for changes in mass loadings, for changes in plant processes and for ocean disposal. In each case, it must be demonstrated that the receiving water quality is not degraded and other existing water uses will not be adversely affected.
- In land development and land use. It must be showed what water quality changes will occur as a result of the proposed development and what effect the land development or land use will have on the existing water uses.
- In the approval process for dams and in their operation. It is required to demonstrate that the construction of the dam will not adversely affect the water quality either upstream or downstream from the dam site. And the operator of the dam must show that the operating procedures will not adversely affect the downstream and upstream water uses.
- To resolve water use conflicts like degraded water quality at a water intake, or for other water uses.
- In the allocation of water resources to different water uses like drinking water, irrigation, fisheries and recreational facilities.
- In the operation of irrigation withdrawals. The prime concern is that the downstream in-flow stream needs to support a viable fishery. Water quality models are extensively used to predict downstream water temperatures and water quality.
- In spill management. Models are used primarily for coastal spills of oil to assist in the allocation of remedial measures.

2.3.2 Modeling approaches

Environmental modeling can be classified into three basic approaches (Nirmalakhandan, 2002):

- Physical modeling or experimental modeling, involves representing the real system by a geometrically and dynamically similar, scaled model and conducting experiments on it to make observations and measurements. The results from these experiments are then extrapolated to the real systems.

- Empirical modeling or black box modeling is based on an inductive or data-based approach, in which past observed data are used to develop relationships between variables believed to be significant in the system being studied. The resulting model is considered a "black box," reflecting only what changes could be expected in the system performance due to changes in inputs.
- Mathematical modeling or mechanistic modeling is based on the deductive or theoretical approach. Here, fundamental theories and principles governing the system along with simplifying assumptions are used to derive mathematical relationships between the variables known to be significant. The resulting model can be calibrated using historical data from the real system and can be validated using additional data. Predictions can then be made with predefined confidence. In contrast to the empirical models, mathematical models reflect how changes in system performance are related to changes in inputs.

Mathematical modeling forms the foundation for computer modeling, while both physical and empirical modeling approaches provide valuable information to the mathematical modeling process.

2.3.3 Mathematical modeling

A mathematical model is a mathematical equation or a set of equations that translates a conceptual understanding of a system or process into quantitative terms (Reckhow and Chapra, 1983).

2.3.3.1 Mathematical modeling classification

In general, water quality models are usually classified according to model complexity, type of receiving water and the water quality parameters (dissolved oxygen, nutrients, etc.) that the model can predict. The more complex the model is, the more difficult and expensive will be its application to a given situation. Model complexity is a function of four factors (World Bank Group. et al., 1999):

- *The number and type of water quality indicators*. The more indicators that are included, the more complex the model will be.
- *The level of spatial detail*. As the number of pollution sources and water quality monitoring points increase, so do the data required and the size of the model.
- *The level of temporal detail*. It is much easier to predict long-term static averages than short-term dynamic changes in water quality.
- *The complexity of the water body under analysis*. Small lakes that "mix" completely are less complex than moderate-size rivers, which are less complex than large rivers, which are less complex than large lakes, estuaries and coastal zones.

Mathematical models can be classified into the following various types depending on the nature of the variables, the used mathematical approaches and the behavior of the system:

- Deterministic or stochastic (probabilistic). Deterministic models use expected values for all parameters and variables and yield predictions that are also expected values. Stochastic models incorporate variability, and possibly error, in probability density functions for selected parameters and/or variables. Deterministic models are built of algebraic and differential equations, while stochastic models include stochastic features (e.g., first-order error analysis or Monte Carlo simulation).

- Continuous or Discrete. When the variables in a system are continuous functions of time, then the model for the system is classified as continuous. If the changes in the variables occur randomly or periodically, then the corresponding model is termed discrete. Continuous models are often built of differential equations; discrete models, of difference equations.
- Static or dynamic. These terms refer to the presence or absence of a time-dependency. Static or steady-state models describe behavior that is constant over time, i.e. time-independent. While dynamic models describe behavior that varies with time and i.e. time-dependent. Static models, in general, are built of algebraic equations resulting in a numerical form of output, while dynamic models are built of differential equations that yield solutions in the form of functions.
- Lumped or distributed. These terms refer to the presence or absence of a space-dependency. Lumped parameter models are zero dimensional in space; they are based on an assumption of uniform condition throughout the system modeled, while distributed parameter models are developed to describe systems with variable conditions in one or more spatial dimension. Lumped, static models are often built of algebraic equations; lumped, dynamic models are often built of ordinary differential equations; and distributed models are often built of partial differential equations.
- Analytical or numerical. When all the model equations can be solved algebraically to yield a closed form solution, the model can be classified as analytical. If that is not possible, and a numerical procedure is required to solve one or more of the model equations, the model is classified as numerical.
- Cross sectional or longitudinal (time series). Cross-sectional models describe behavior among cases, in contrast to longitudinal models, which describe behavior for a single case over time. Cross-sectional models are more often empirical, whereas longitudinal models may be either empirical or mathematical (mechanistic).

2.3.3.2 Mathematical modeling developing

The development of mathematical models should proceed according to certain general rules. When a model is to be developed to aid water quality management planning, the following steps are suggested:

1. Problem formulation. This step involves the following tasks:

- Task 1: determining objectives and establishing criteria for meeting the objectives.
- Task 2: characterizing the system by developing a conceptual model of the system of interest and of the relevant attributes of the system (e.g., the boundary, the system-surroundings

interactions, the directions and the rates of flows crossing the boundary and reaction and process rates inside the system).

- Task 3: simplifying and idealizing the system by making appropriate assumptions and approximations, based on the available resources and the system characteristics, to simplify the system, to make it amenable to modeling.

2. Mathematical representation. Constructing the mathematical model from the conceptual model by using the following tasks:

- Task 1: identifying the fundamental theories (e.g., conservation of mass, reaction theory and transport mechanisms).
- Task 2: deriving relationships between the variables of significance and relevance.
- Task 3: standardizing relationships by reducing them to standard mathematical forms.

3. Mathematical analysis. Applying the standard mathematical techniques and procedures (analytical or numerical) to solve the model to obtain the desired results.
4. Interpretation and evaluation of results. This process consists of two main tasks:

- Task 1: calibrating the model by using previously observed data set from the real system ("training" set). The model is run repeatedly, adjusting the model parameters by trial and error (within reasonable ranges) until the best values for the parameters of the model are estimated (along with error terms).
- Task 2: validating (or confirming) the model by using a "testing" data set from the real system (different from those used to calibrate). The calibrated model is run under conditions similar to those of the testing set, and the results are compared against the testing set. The accuracy of the model, determined by validation, is affected by two types of uncertainty: uncertainty in the model and uncertainty in the reference data. In most ecological models the uncertainty in the reference data is very large, and this propagates through the calculations to give a poor accuracy of the model (Dahl and Wilson, 2001).

5. Applying the model as intended.

The overall approach in mathematical modeling is illustrated in Figure 2.29.

2.3.3.3 Data limitation

Lack of data can create three problems (World Bank Group. et al., 1999):

1. A model cannot be calibrated or tested until a monitoring system has been designed and operated for a considerable length of time.
2. Water sample collection and analysis may be considerably more expensive than the modeling effort that it is designed to support.
3. Design of a monitoring system may fall prey to the same types of problems that can affect water quality modeling, including a lack of clear connections to management objectives and a tendency to excessive complexity.

2.3.3.4 Mathematical modeling state of the art

In the 1920s the Ohio River Commission, USA, began an intensive study of sources of pollution and their impacts on domestic water supply. This investigation emerged one of the first mathematical models of an aquatic environment, the Streeter- Phelps equation which describes the balance of dissolved oxygen in a stream (Streeter and Phelps, 1925). It was stated as a first-order ordinary differential equation that, when integrated, provided an analytical solution describing the dissolved oxygen deficit as a function of time from the point of discharge of a steady source of biodegradable pollutant, this solution called the "oxygen sag equation".

Jorgensen et al. (1996) and Jorgensen and Bendoricchio (2001) display the development of ecological modeling in five different generations:

1. The first generation of models was in the 1920s with the Streeter- Phelps model.
2. The second generation was in the 1950s and the 1960s where more complex river models were developed.
3. The third generation (computer technology generation) started around the year 1970, when the first eutrophication models emerged and very complex river models were developed and extended to the mid-1970s.
4. The fourth generation started from the mid-1970s to the mid-1980s and its models are characterized by having a relatively sound ecological basis, with emphasis on realism and simplicity.
5. The fifth generation has started since the mid-1980s where modelers have proposed many new approaches.

Thamoann (1998) states that the evolution of models is seen in three stages. During the first stage (1925 to about 1980), all sources (point, nonpoint and sediment) were external to the model, but only point sources were directly linked to the originating input. During the second stage (about 1980 to 1990), sediment models were coupled to the water column and hydrodynamic and watershed models were linked. A link was then established from watershed models to the input of the watershed. During the third stage (currently under way), airshed models are being incorporated with expansion to include other aspects of aquatic ecosystem.

Chapra (1997) shows that the historical development of water quality models can be broken down into four major phases, as seen in Figure 2.30. These phases relate to both the societal concerns and the computational capabilities that were available during each of the periods.

Orlob (1983a) provides state-of-the-art overview of mathematical modeling of lakes and reservoirs starting (about 1968) with one-dimensional reservoir models which developed to simulate the vertical distribution of heat in the impoundment over an annual cycle.

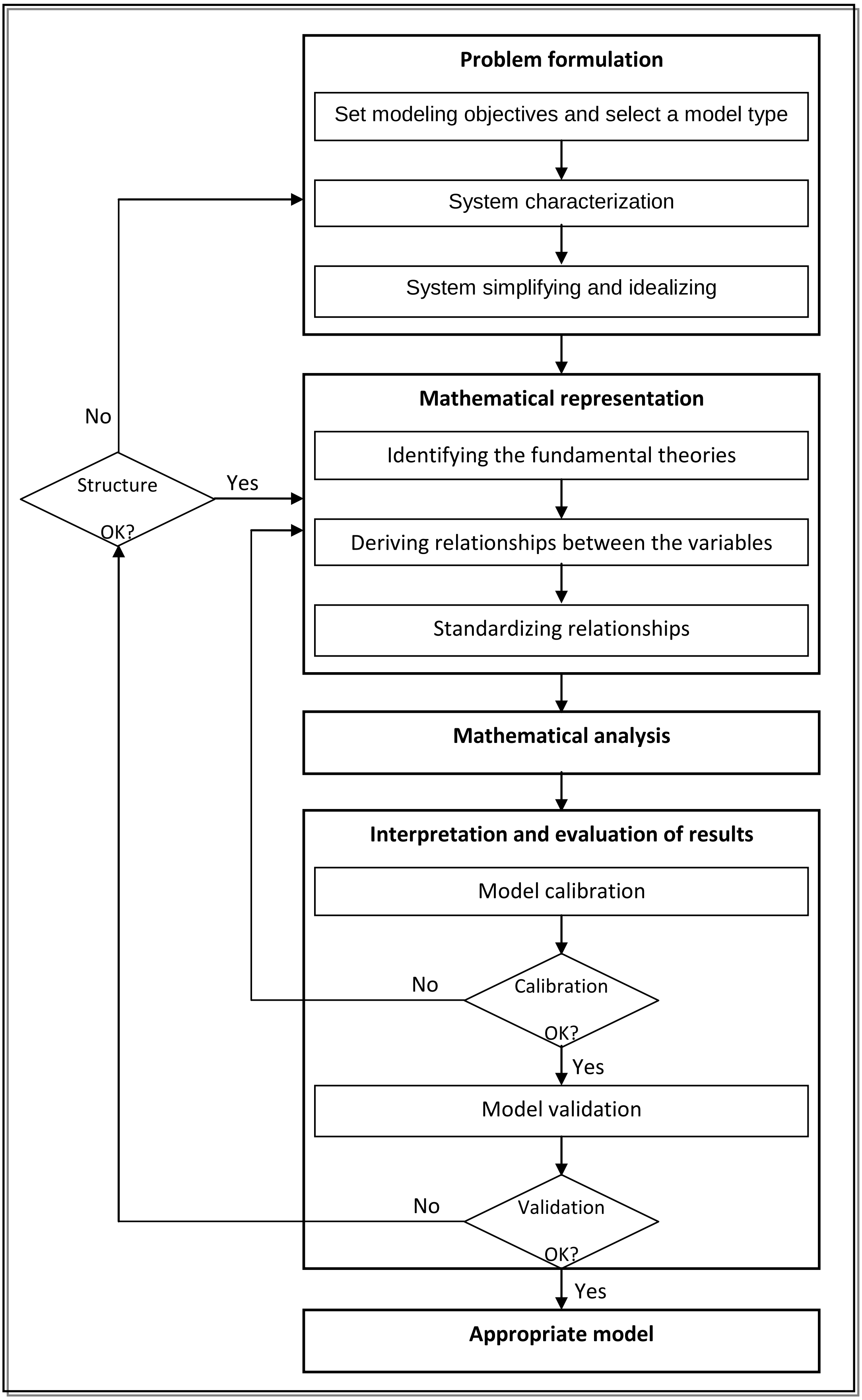

Figure 2.29: Modeling methodology, modified after Dahl and Wilson (2001).

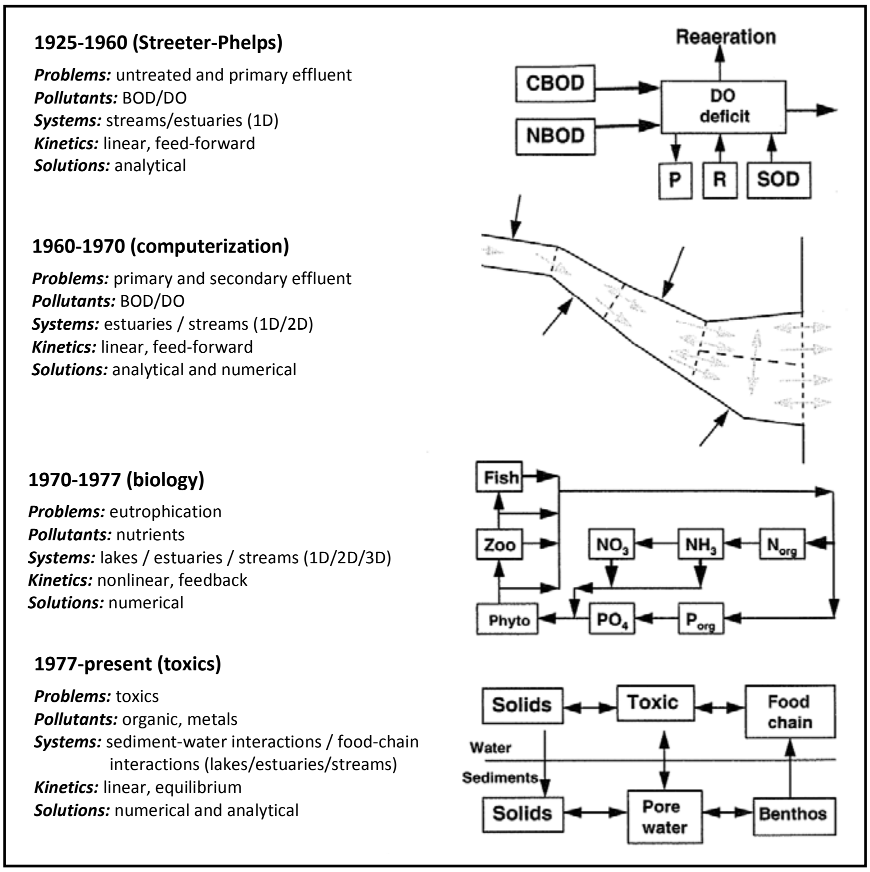

Figure 2.30: Four periods in the development of water quality modeling (Chapra, 1997).

2.3.4 Hydrodynamics and transport

Hydrodynamics studies the motion of water and the forces acting on water and it is the driving mechanism for transport of sediments, toxics and nutrients (Ji, 2008).

2.3.4.1 Fundamental hydrodynamic processes

2.3.4.1.1 Laws of conservations

Laws of conservations are derived from Newtonian mechanics. They form the theoretical basis of hydrodynamics and are used routinely in the studies of hydrodynamics and water quality. Laws of conservations include (i) the conservation of mass, (ii) the conservation of energy and (iii) the conservation of momentum.

Conservation of mass

The law of conservation of mass states that mass can neither be created nor destroyed, but merely transferred or transformed. This law forms the basis for transport models which are referred to as water quality models. It is often expressed in a mass balance equation (also called continuity equation), which accounts for the flux of mass going into a defined area and the flux of mass leaving the defined area (Eq. 2.2).

$$Mass\ accumulation = Mass\ in - Mass\ out + Sources - Sink \tag{2.2}$$

Conservation of energy

The law of conservation of energy states that energy associated with matter entering any system plus the net energy added is equal to the energy leaving the system, or the net work done by the system, and the change in energy within the system. This law forms the basis for temperature models. It is used mainly to formulate a heat balance for temperature and evaporation modeling and to derive useful mixing relationships from the conservation of total kinetic energy.

Conservation of momentum

The law of conservation of momentum states that momentum can neither be created nor destroyed, but merely transferred or transformed. It was first noted by Galileo and forms the basis of Newton's first law of motion. This law forms the basis for flow models (hydrodynamic and hydraulic). The conservation of momentum can be derived from Newton's second law (Eq. 2.3):

$$F = m \cdot a \tag{2.3}$$

where:

F	External force (M L T^{-2})
m	Mass of the object (M)
a	Acceleration of the object (L T^{-2})

For surface water bodies, F represents the following forces:

1. Gravitational force.
2. Force from water pressure gradient.
3. Viscous force.
4. The Coriolis force (the effect of earth rotation on water movement).
5. Wind forces.

The momentum equation stated in equation (2.3) is called the Navier-Stokes equation, which has no analytical solutions and is also too complex to be solved numerically for large domains over long periods of time unless further simplifications are applied to it.

2.3.4.1.2 Hydrodynamic mass transport

Hydrodynamic transport is a vital process that moves materials from the location at which they are generated, resulting in impacts that can be distant from the material source due to the water movement. Materials in water systems can be transported by one or all of the following processes: (1) advection, (2) dispersion and (3) vertical mixing and convection.

Advection refers to horizontal transport by flow that moves a property from one location to another without distorting or diluting it. In contrast to advection, *convection* refers to vertical transport of water and pollutants. Convection in rivers, lakes and estuaries is usually very small.

Dispersion is the horizontal spreading and mixing of water mass caused by *turbulent mixing* and *molecular diffusion*. Dispersion reduces the gradient of material concentration. This process involves not only an exchange of water mass, but also of any substance dissolved in it, such as salinity and dissolved pollutants. Dispersion in the direction of water flow is called longitudinal dispersion and perpendicular to the direction of flow is called lateral dispersion.

Figure 2.31 shows a pictorial representation of the transport of a dye patch in space and time via (a) advection and (b) Diffusion (Chapra and Reckhow, 1983).

Turbulent mixing is often the dominant component of dispersion in rivers, lakes and estuaries and is much more rapid than molecular diffusion. Turbulent mixing is the result of the momentum exchange between water parcels in a turbulent flow. It spreads chemical or biological constituents in various directions depending on the flow characteristics.

Diffusion is a transport process at the microscopic level, owing to the scattering of particles by random molecular motions. Diffusion is the movement of materials from an area of high to an area of low concentration due to the concentration gradients.

Advection and dispersion are the major processes by which dissolved materials are transported along and distributed throughout a river or an estuary. As water flows along the river, it transports dissolved materials with it via advection. It leads to a net transport of dissolved materials from areas of high concentration to areas of low concentration via dispersion. Hence, horizontal transport of a material consists of two components: advective flux and dispersive flux (Ji, 2008).

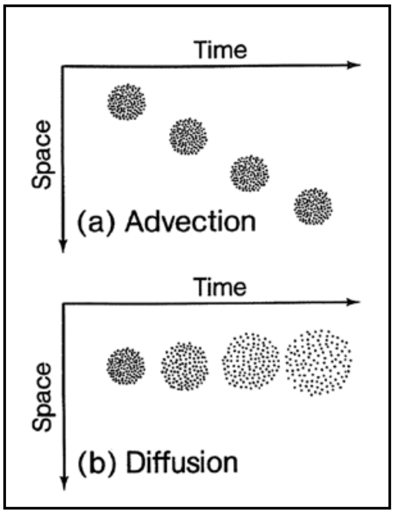

Figure 2.31: a pictorial representation of the transport of a dye patch in space and time via (a) advection and (b) Diffusion (Chapra and Reckhow, 1983).

Advective flux density J_a ($M\ T^{-1}\ L^{-2}$) – in the flow direction – can be determined from Eq. (2.4):

$$J_a = C \cdot v \tag{2.4}$$

where:

C Material concentration in the water ($M\ L^{-3}$)
v Flow velocity ($L\ T^{-1}$)

Dispersive mass flux density J ($M\ T^{-1}\ L^{-2}$) – from areas of high concentration to areas of low concentration – can be determined by using Fick's Law which states that the rate of mass movement resulting from molecular diffusion is inversely proportional to the gradient of mass concentration:

$$J = -D\frac{dC}{dx} \tag{2.5}$$

where:

D	Diffusion coefficient (L^2 T^{-1})
C	Material concentration in the water (M L^{-3})
x	Distance (L)

Equation (2.5) determines the dispersive mass flux density due to molecular diffusion. Turbulent mixing can be considered roughly analogous to molecular diffusion. Hence, the dispersive mass flux density due to turbulent mixing also follows Fick's Law (Eq. 2.5), only that the magnitude of the diffusion coefficient (*D*) is different.

The total mass flux across a boundary can be calculated as:

$$\frac{dm}{dt} = (J_a + J)A \tag{2.6}$$

where:

m	Mass (M)
A	Area of the boundary that perpendicular to the direction of the flow (L^2)

In most natural water bodies, the advective flux (J_a) is larger than the dispersive flux (J). When the flow velocity is very small, the advection flux becomes small and can be neglected. Hence, the conservation of mass can then be simplified as:

$$\frac{\partial C}{\partial t} = -\frac{\partial J}{\partial x} \tag{2.7}$$

Combining equations (2.4) and (2.6) yields;

$$\frac{\partial C}{\partial t} = D\frac{\partial^2 C}{\partial x^2} \tag{2.8}$$

Equation (2.8) is the classical diffusion equation from Fick's Law. Its solution needs one initial condition and two boundary conditions.

2.3.4.1.3 Reaction kinetics

Contaminants can be transported through a system by microscopic and macroscopic mechanisms such as diffusion, dispersion and advection. At the same time, they may or may not undergo a variety of physical, chemical and biological processes within the system. Some of these processes result in changes in the molecular nature of the contaminants. These processes are referred as *reactive processes* (e.g. biodegradation, oxidation and reduction) and the contaminants which undergo these processes are called *reactive substances*. The other processes (e.g. settling, resuspension, flotation and filtration) are *nonreactive processes* and their contaminants are called *conservative substances*.

The reactive processes can be described by reaction equations which should be incorporated in the mass balance equation. The *kinetics* or speed of such reactions can be expressed quantitatively by the Law of Mass Action, which states that the rate is proportional to the concentration of the reactants (Chapra and Reckhow, 1983). Consider the reaction: $A + B \rightarrow C$. The rate of consumption of species A by this reaction is given by the Law of Mass Action as follows (Nirmalakhandan, 2002):

$$\frac{dC_A}{dt} = -kf(C_A\,, C_B) \tag{2.9}$$

where:

C_A	Concentration of A (ML^{-3})
k	Reaction rate coefficient which depends on a number of factors including temperature and the chemical characteristics of the reacting substances
$f(C_A\,, C_B)$	Function has to be determined experimentally

A common general form of this equation is

$$\frac{dC}{dt} = -kC^n \tag{2.10}$$

where n is the reaction order, n = 0, 1, 2 for zero-, first- and second-order reaction respectively.

The rate of reaction *(dC/dt)* can be either positive or negative. For example, the degradation of one chemical can serve as a source for another, such as nitrification of ammonia being a source for nitrate-nitrogen (Martin et al., 1999).

Among the most used processes equations is the Michaelis-Menten formulation originally set for biochemical enzyme kinetics (Somlyódy and Varis, 1992),

$$\frac{dC}{dt} = \frac{k_1}{k_2 + C}C \tag{2.11}$$

where k_1 defines the maximum value of $\frac{dC}{dx}$ (i.e. the maximum grows rate) and k_2 is the half-saturation constant. If *C* is small, equation (2.11) behaves like a first order reaction, while for large *C* as a zero order one.

2.3.4.1.4 Temperature simulation (heat balance)

Water temperature is a function of both surface heat flux and the transport of water into and out of the system. The total heat budget for a water body includes the effects of heat exchanges with the atmosphere and with the water bottom, inflows/outflows and heat generated by chemical/biological reactions. The dominant process controlling the heat budget is the atmospheric heat exchange (Ji, 2008).

Mainly the heat exchange between the atmosphere and the water columns are transferred by:

1. Radiation processes, including:

- The solar radiation. It is the short-wave radiation from the sun that reaches the water surface.
- The long-wave radiation. It is the net long-wave radiation from the atmosphere and the water body.

2. Turbulent heat transfers, including:

- The latent heat. It is the heat transfer due to water evaporation.
- The sensible heat. It is the conductive heat transfer between the atmosphere and the water body.

Figure 2.32 summarizes the major four heat flux components. In addition to them, there are other minor heat sources (negligible in most applications) including:

1. Heat generated from chemical/biological reactions.
2. Heat exchange between the water and the water bottom.
3. Heat generated from current friction.

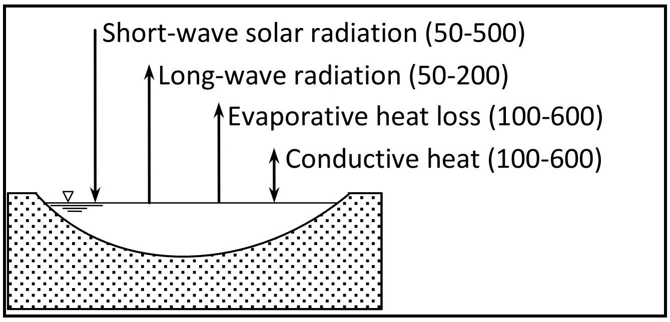

Figure 2.32: Components of surface heat exchange along with representative ranges in units of Watt/m^2, modified after Martin et al. (1999).

Solar radiation

Often the short-wave solar radiation is the most important heat flux component in terms of magnitude. A variety of empirical formulas are proposed to estimate the solar radiation. Unlike the three other heat flux components, which all occur only at the water surface, solar radiation can penetrate to considerable depths, depending on the color and turbidity of the water. The relationship of radiation remaining at a given depth ($H_{sw,z}$), to the solar radiation at the water surface (H_{sw}) can be described using Beer's Law (Eq. 2.12).

$$H_{sw,z} = H_{sw} \cdot e^{-k_e z} \tag{2.12}$$

where:

$H_{sw,z}$	Solar radiation (M L^2 T^{-3}) at given depth z (L)
H_{sw}	Solar radiation at the water surface (M L^2 T^{-3})
k_e	Light extinction coefficient determined for a particular water body (L^{-1})

Another effect is the shading from trees and steep riverbanks can significantly reduce the incoming solar radiation to the water surface, resulting in water temperatures much lower than those in unobstructed areas. When modeling narrow rivers and reservoirs, the shading effect can be significant for accurately simulating the local water temperature.

Long-wave radiation

The difference in solar radiation at the top of the atmosphere versus that at the water surface is the radiation absorbed by the clouds and the atmosphere. This atmospheric heat is in turn reflected at longer wave lengths and is considered the greatest source of heat at the water surface on cloudy days. Water also emits long-wave radiation, which represents a loss of heat. The amount of heat loss is generally described using the Stefan-Bolzmann Law.

The net long-wave radiation on the water-air surface is the result of the downward radiation from the atmosphere and the upward radiation emitted by the water surface.

Evaporation and latent heat

Evaporation is a cooling process by which water at the water surface is converted from the liquid to the vapor state. The latent heat due to evaporation is the major heat loss for a water body. Various theoretical and empirical formulas have been proposed to estimate the latent heat flux. A common approach is to link the latent heat flux with wind speed and the difference between the saturated vapor pressure at the water surface temperature and the actual vapor pressure in the overlying air.

Evaporation can be a significant component in water balance, especially for subtropical lakes where evaporation rate is very high due to high water temperatures. Evaporation can be calculated using:

$$H_L = \rho L_w E \tag{2.13}$$

where:

H_L	Latent heat flux due to evaporation (M $L^2 T^{-3}$)
ρ	Water density (M L^{-3})
L_w	Latent heat of water (M $L^2 T^{-2}$)
E	Evaporation rate (L T^{-1})

Sensible heat

Sensible heat exchange (or conduction) is the heat flux transferred between water and the atmosphere due to a temperature difference between the two. The heat exchange takes place by conduction and convection and occurs only in a very thin layer of the air – water boundary. Various empirical formulas have been proposed to estimate the sensible heat flux.

The net heat flux for a water body can be described by:

$$H_{net} = H_{sw} + H_{Lw} + H_L + H_s \tag{2.14}$$

where:

H_{net}	Net heat flux across the air/water interface (M $L^2 T^{-3}$)
H_{sw}	Short-wave solar radiation flux (M $L^2 T^{-3}$)
H_{Lw}	Net long-wave radiation flux from the atmosphere and the water body (M $L^2 T^{-3}$)
H_L	Latent heat flux due to evaporation (M $L^2 T^{-3}$)
H_s	Sensible heat flux due to conduction (M $L^2 T^{-3}$)

2.3.4.2 Hydrodynamic processes for lakes and reservoirs

2.3.4.2.1 Inflow

Inflows contribute to the mixing of lakes and reservoirs and are considered the primary source of dissolved and particulate materials.

If the kinetic energy due to inflow is large or the density gradient (between the inflow water and the lake water) is small, inflow can cause a lake to completely mix. If there is a density difference between the inflow water and the lake water, then the inflow will flow to a layer of equivalent density level and more along that layer. Density differences result from differences in temperature or dissolved materials *(density currents)* or differences in particulate loads *(turbidity currents).*

When river waters are warmer than lake waters, the inflow waters will tend to spread out over the lake surface in the form of overflow (Figure 2.33). Mixing between the lake waters and the inflow waters will increase as temperature difference is reduced. Overflows are common in the springtime. Overflows can contribute to water quality variations, as they add materials directly to the more productive surface zones of the lake.

When surface waters are cooling down, river waters are often colder than lake waters, and the inflow will drop below the lake surface and submerge to a depth where the densities of the lake and that of the incoming waters are equal. As a result, the inflow forms a density flow below the warmer surface water, but above the colder bottom water forms an interflow (Figure 2.33). Or the inflow sinks all the way to the bottom of the lake and forms an underflow (Figure 2.33). Underflows are common in the autumn.

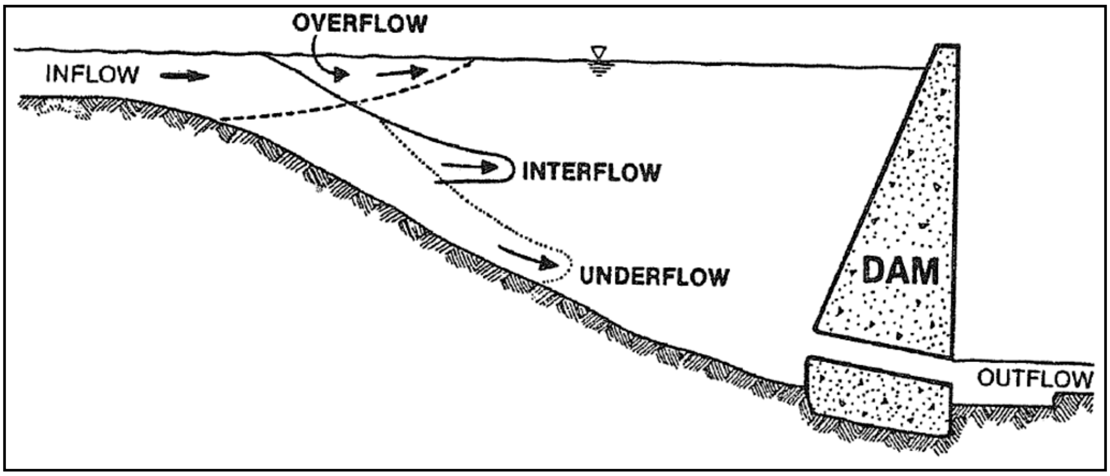

Figure 2.33: Density currents in lakes and reservoirs (Martin et al., 1999).

Plunge point is the location at which the denser inflow plunges beneath the water surface and becomes a density current (Figure 2.34). Due to the velocity shear between the underflow and the overlying lake water, some of the lake water is dragged downwards, and a corresponding counter flow is induced in the upper layer of the lake. Because of this surface velocity pattern, the plunge point is sometimes marked by floating debris on the lake surface.

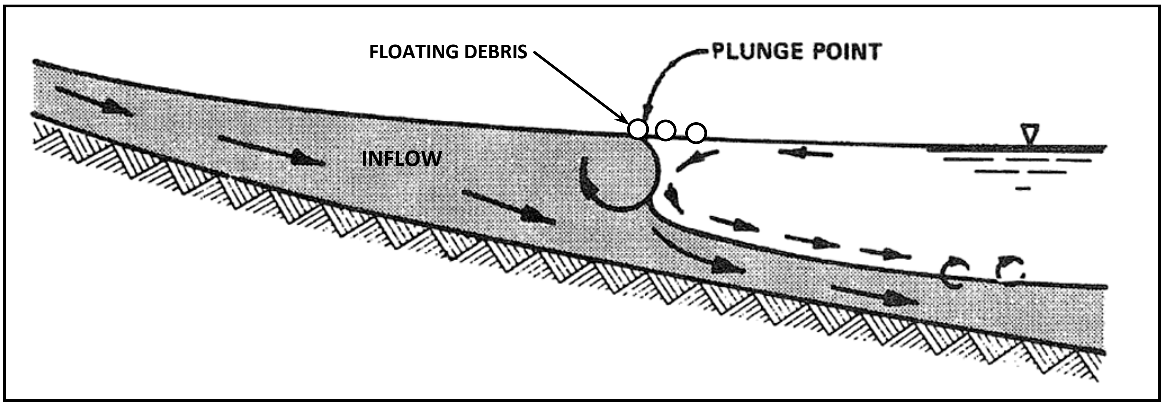

Figure 2.34: Mixing at the plunge point, modified after Martin et al. (1999).

Martin et al., 1999, state useful relationships to estimate the location of plunge point, thickness and width of overflow, thickness and length of interflow and thickness of underflow.

2.3.4.2.2 Outflow

Outflows include natural releases from lakes and discharges at reservoir dams. Natural lakes often have discharges from the lake surface. For reservoirs, however, discharges are normally regulated by passing through control structures of the dam. When water is released from a reservoir, potential energy is converted into kinetic energy. Mixing is a result of this conversion of energy, and the degree of mixing varies with the location of the discharge outlets. As water is released from a reservoir, a withdrawal zone develops (Figure 2.35). The extent of this zone depends on the release rate, the location of the withdrawal, reservoir bathymetry and the lake stratification. The withdrawal velocities can be used directly in models to predict the effects of withdrawals on reservoir water quality.

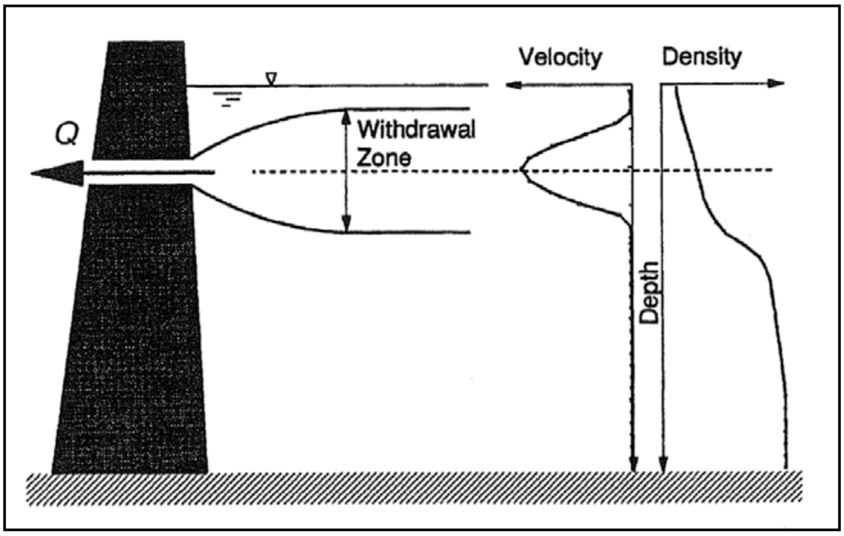

Figure 2.35: Typical withdrawal pattern (Martin et al., 1999).

2.3.4.2.3 Wind forcing

There are three external forcing are essential to the hydrodynamic processes in lakes and reservoirs:

- Heat flux exchanges and thermal forcing.

- Inflow and outflow.
- Wind forcing.

Wind forcing determines lake circulation and the major energy source for vertical mixing. When the wind blows across a lake, it affects the lake as follows (Ji, 2008):

- It exerts a shear stress on the water surface.
- It results in momentum transfer from the air into the water.
- It causes the surface water to move in the direction of the wind.

Wind energy is converted into turbulence in the surface layer. Then it is transferred to the lower parts of the epilimnion by turbulent diffusion, until the thermal gradient dissipates the energy. In shallow lakes wind-induced turbulence may occur at all depths, and therefore can significantly enhance nutrient entrainment from the sediment bed. As the water depth increases, however, this wind-induced turbulence cannot reach the bottom except around the lake edges. In such deep lakes, sediment re-suspension is weak and nutrients tend to accumulate on the bed.

2.3.4.2.4 Coriolis force

The rotation of the earth impacts water movement in lakes and reservoirs. Coriolis force, due to the rotation of the earth, deflects currents to the right (left) in the northern (southern) Hemisphere. Coriolis force is significant only when the spatial scale of the study area is very large. When the frictional forces are neglected, the steady-state flow is determined by the balance between the pressure gradient force and the Coriolis force, this balance is known as geostrophic flow.

2.3.4.2.5 Langmuir circulation

In very large lakes with a very long fetch, a series of parallel pairs of large vertical vortices or circulatory cells known as Langmuir circulation develop (Figure 2.36), in the general direction of a sustained wind when wave and current conditions are favorable. These long vortices develop side by side over the entire surface of the deeper water but a slight angle to the wind direction. Langmuir circulation contribute to the vertical mixing of the epilimnion (Martin et al., 1999).

2.3.4.2.6 Gyres

A gyre is a circular, rotational circulation pattern, established by winds or other physical forces in large shallow lakes (Figure 2.37). When a uniform wind blows over a large and shallow lake that is shallower on the right and deeper on the left, the line of action of the wind forcing is through the centroid of the water surface. Since it is deeper and contains more water on the left, the mass center of the lake water should be toward the deeper side, to the left of the line of centroid. Therefore, the mass center and the line of centroid do not coincide, and a torque is produced. The torque makes the lake water rotate, flow into the paper on the right and flow out of the paper on the left.

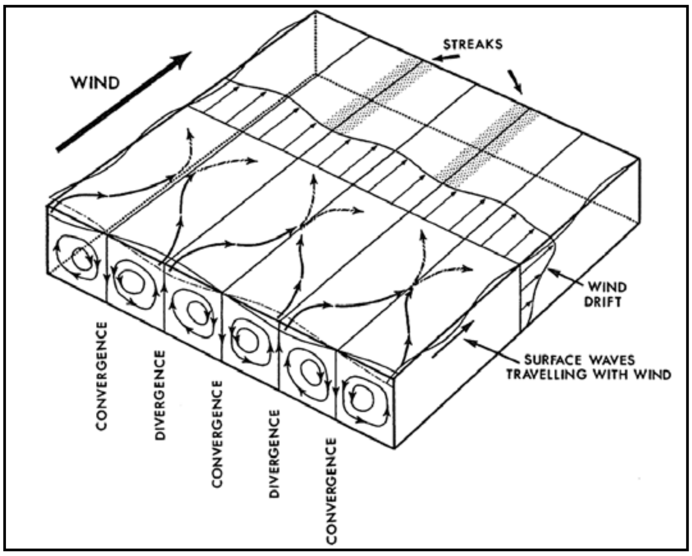

Figure 2.36: Diagrammatic representation of the helical flow of Langmuir circulations in surface waters with streaks of aggregated organic matter occurring at lines of divergence (Wetzel, 2001).

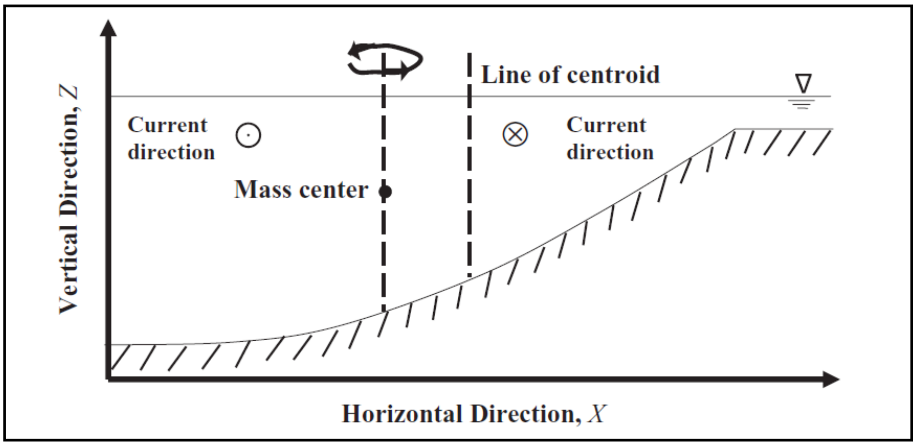

Figure 2.37: A sketch on gyre formation caused by wind forcing, modified after Ji (2008).

2.3.4.2.7 Seiches

In lakes and reservoirs, internal waves are more important than surface waves for mixing. Internal waves are produced by wind forcing, withdrawals, hydropower releases, thermal discharges and local disturbances. One of the most important internal waves is the seiche. Seiches are standing waves (or periodic oscillations) of the water level in closed or semi-closed water bodies, such as lakes. Prolonged wind forcing on a lake produces a surface gradient in which the water level rises in the downwind sector (is known as wind setup). Oscillations take place when the wind forcing suddenly reduces or changes direction. Because the solid barrier of the lake boundary reflects waves, the superposition of the original and reflected waves gives rise to standing waves called seiches.

2.3.5 Theoretical approach

Most of lake and reservoir water quality models include the following components:

- A water balance or water continuity equation.

- A series of mass balance or advective-dispersive equations, one mass balance equation for each water quality constituent of interest.
- Momentum equations in one, two or three directions.

2.3.5.1 Basic approximations

Laws of conservations provide the fundamental principles needed to develop hydrodynamic models. Even with advanced computers, these conservation equations are too complex to be solved numerically for large domains over long periods of time. Therefore, further simplifications are needed.

Shallow water approximation

Shallow water approximation is a widely used approximation in surface water studies. It states that horizontal scale of motion, *L,* is much larger than the vertical scale of motion, *H*. This approximation is justified for most hydrodynamic processes in rivers, lakes, estuaries and coastal waters. It is often assumed to be valid when $H/_L \leq 0.05$ (Ji, 2008).

Boussinesq approximation

Boussinesq approximation assumes that the water is incompressible, this means that the water density does not change with water pressure. So variations in water density are ignored, except when the gravitational force and buoyancy are considered. Density changes due to local pressure gradients in the horizontal momentum equations are negligible.

Hydrostatic approximation

The hydrostatic approximation assumes that the vertical pressure gradient is almost balanced by the forcing due to buoyancy excess. The vertical acceleration then is a much smaller term and can be omitted from Navier-Stokes equation. When horizontal scales are much greater than vertical scales, the hydrostatic approximation is justified and in fact, is identical to the shallow water approximation for continuously stratified waters. Most of the 2D (vertical plane) and 3D hydrodynamic models use this approximation.

Quasi-3D approximation (two-and-a-half-D or pseudo-3D)

An alternative to develop a fully 3D model is to treat the system as a set of horizontal layers that interact via source–sink terms representing water exchanges with overlying and underlying layers. This approach allows for eliminating the momentum equation in the vertical direction. For most surface water applications, the quasi-3D approximation ensures computational efficiency and model accuracy. Most 3D hydrodynamic models used in rivers, lakes and estuaries are actually quasi-3D models. Also in most 2D laterally averaged models a similar approach is also used so that the vertical momentum equation is not computed.

2.3.5.2 Governing equations

To develop the general complete equations set of water quality models, the Navier-Stokes equations for the conservation of momentum and the conservation laws for mass and energy are required (Watanabe et al., 1983).

The three components of velocity, *u, v, w and* the pressure *p* are decomposed according to the Reynolds conceptualization into mean and fluctuating components:

$$u = \bar{u} + \grave{u}, \qquad v = \bar{v} + \grave{v}, \qquad w = \bar{w} + \grave{w}, \qquad p = \bar{p} + \grave{p}.$$

By introducing the Boussinesq and the hydrostatic approximations in the Navier-Stokes equations and the time averages are taken, the Reynolds equations for a fluid body on a rotating earth are as follows:

$$\frac{\partial \bar{u}}{\partial t} + \frac{\partial}{\partial x}\bar{u}\bar{u} + \frac{\partial}{\partial y}\bar{u}\bar{v} + \frac{\partial}{\partial z}\bar{u}\bar{w} - \Omega\bar{v} = -\frac{1}{\rho}\frac{\partial \bar{p}}{\partial x} + \nu\nabla^2\bar{u} - \frac{\partial}{\partial x}\overline{\grave{u}\grave{u}} - \frac{\partial}{\partial y}\overline{\grave{u}\grave{v}} - \frac{\partial}{\partial z}\overline{\grave{u}\grave{w}} \quad (2.15)$$

$$\frac{\partial \bar{v}}{\partial t} + \frac{\partial}{\partial x}\bar{v}\bar{u} + \frac{\partial}{\partial y}\bar{v}\bar{v} + \frac{\partial}{\partial z}\bar{v}\bar{w} - \Omega\bar{u} = -\frac{1}{\rho}\frac{\partial \bar{p}}{\partial y} + \nu\nabla^2\bar{v} - \frac{\partial}{\partial x}\overline{\grave{v}\grave{u}} - \frac{\partial}{\partial y}\overline{\grave{v}\grave{v}} - \frac{\partial}{\partial z}\overline{\grave{v}\grave{w}} \quad (2.16)$$

$$0 = -g - \frac{1}{\rho}\frac{\partial \bar{p}}{\partial z} \quad (2.17)$$

where:

t	Time (T)
x, y, z	Coordinate axes (*+z* is vertically upward) (L)
$\bar{u}, \bar{v}, \bar{w}$	Mean velocities in the *x, y, z* directions (L T^{-1})
$\overline{\grave{u}_i \grave{u}_j}$	Time-averaged turbulent eddy transport of momentum
g	Gravitational acceleration (L T^{-2})
ρ	Density (M L^{-3})
$\bar{p}$	Pressure (M L^{-1} T^{-2})
Ω	Coriolis parameter (T^{-1})
ν	Molecular kinematic viscosity (M L^2 T^{-1})
∇^2	$= \partial^2/\partial x^2 + \partial^2/\partial y^2 + \partial^2/\partial z^2$

Similarly, the equation of heat balance for a turbulent flow is

$$\frac{\partial \bar{T}}{\partial t} + \frac{\partial}{\partial x}\bar{T}\bar{u} + \frac{\partial}{\partial y}\bar{T}\bar{v} + \frac{\partial}{\partial z}\bar{T}\bar{w} = \chi\nabla^2 T - \frac{\partial}{\partial x}\overline{\grave{T}\grave{u}} - \frac{\partial}{\partial y}\overline{\grave{T}\grave{v}} - \frac{\partial}{\partial z}\overline{\grave{T}\grave{w}} + Q_H \quad (2.18)$$

where:

χ	Molecular thermal diffusivity
$\overline{\grave{T}\grave{u}_i}$	Time-averaged turbulent eddy transport of heat in *x* direction
Q_H	Heat source

Advective transport $\bar{T}\bar{u}_i$ and turbulent eddy transport $\overline{\grave{T}\grave{u}_i}$ depend on the state of flow.

In addition, an equation of state defines the relationship between temperature and density:

$$\rho = \rho(T) \quad (2.19)$$

Based on the law of conservation of mass, the concentration change of a substance in the water can be determined using mass balance equation (2.2), which is simply an accounting of mass inputs, outputs, reactions and net change. Its 1D form can be simplified as (Ji, 2008):

$$\frac{\partial C}{\partial t} = -U\frac{\partial C}{\partial x} + \frac{\partial}{\partial x}\left(D\frac{\partial C}{\partial x}\right) + S + R + Q \quad (2.20)$$

Or in a conceptual form;

$$\textit{Net change of concentration} = \textit{Advection effect} + \textit{Dispersion effect} + \textit{Settling effect} + \textit{Reactivity effect} + \textit{External Loads} \quad (2.21)$$

where:

C	Substance concentration (M L^{-3})
U	Advection velocity in x direction (L T^{-1})
D	Mixing and dispersion coefficient (L^2 T^{-1})
S	Sources and sinks due to settling and resuspension
R	Reactivity of chemical and biological processes
Q	External loadings to the aquatic system from point and nonpoint sources

When the reactions and the inflow/outflow are neglected and applying the Boussinesq approximation in the mass balance equation, the continuity equation can be expressed as

$$\frac{\partial \bar{u}}{\partial x} + \frac{\partial \bar{v}}{\partial y} + \frac{\partial \bar{w}}{\partial z} = 0 \quad (2.22)$$

Initial conditions and boundary conditions have to be specified for the six unknowns $\overline{u_\iota}, \overline{v_J}, \overline{w_k}, \rho, \bar{p}$ *and* $\bar{T}$. Also, the eddy transport terms $\overline{\grave{u_\iota}\grave{u_J}}$ and $\overline{\grave{T}\grave{u_\iota}}$ need to be parameterized through the flow state variables (Watanabe et al., 1983).

2.3.5.3 Spatial classification of water quality models

Spatially, lake and reservoir models are divided into four main categories:

1. Zero-dimensional models or whole lake.
2. One-dimensional models, vertically layered or in the longitudinal direction.
3. Two-dimensional models, laterally averaged or vertically averaged.
4. Three-dimensional models.

2.3.5.3.1 Zero-dimensional models

The response of lakes to the input of contaminants can sometimes be estimated by assuming that the lake is well mixed. This approximation is justified when (1) wind-induced circulation is strong and (2) the time scale of the analysis is sufficiently long (on the order of a year) that seasonal mixing processes yield a completely mixed lake (Chin, 2006). The well mixed lake is treated as an interconnected system of continuously stirred tank reactor (CSTR) in which it is assumed that all reactions occur instantaneously in a homogeneous medium (Orlob, 1984). These types of models are referred to as zero-dimensional models, input-output models, box models and alike (Stefan et al., 1989).

Chapra and Reckhow (1983) and Thomann and Muller (1987) provide the governing equations of a completely mixed model and their solutions analytically and numerically.

The mass balance equation for the completely mixed model (Figure 2.38), for a finite period of time, can be expressed as:

$$[accumulation] = [inputs] - [outputs] \pm [reactions] \tag{2.23}$$

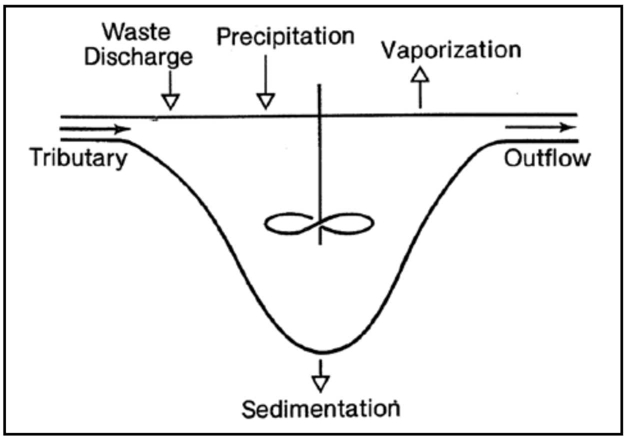

Figure 2.38: Schematic of a mass balance around a completely mixed lake showing some typical inputs and outputs of matter (Chapra and Reckhow, 1983).

Accumulation of a lake of constant volume is the rate of change of a mass of a substance, and can be represented as:

$$\frac{dm}{dt} = V\,\frac{dc}{dt} \tag{2.24}$$

where:

m	Mass of the substance (M)
d/dt	Change with respect to time (T^{-1})
V	Lake's volume (L^3)
c	Concentration of the substance ($M\ L^{-3}$)

Inputs, or loadings, may consist of mass carried by tributary streams, in precipitation, from sediment feedback or from municipal sewer systems. Inputs generally can be characterized by their temporal characteristics. This is expressed mathematically as:

$$\left(V\,\frac{dc}{dt}\right)_{inputs} = W(t) \tag{2.25}$$

where:

$W(t)$	Rate of mass loading ($M\ T^{-1}$)
(t)	Designates it as a function of time

Outputs could be via sedimentation, vaporization or flow through the lake's outlet. Since the flow through the outlet is advective, it can be formulated as:

$$\left(V\,\frac{dc}{dt}\right)_{outflow} = -Qc \tag{2.26}$$

Reaction can transform the substance, which is modeled, into another compound. Reaction can be expressed as:

$$\left(V\,\frac{dc}{dt}\right)_{reaction} = -kVc^n \tag{2.27}$$

where:

k Reaction's coefficient (T^{-1} $(M\ L^{-3})^{1-n}$)
n Order of the reaction

The most common formulation used to characterize reactions in natural waters is the first-order (i.e., $n = 1$) reaction,

$$\left(V\ \frac{dc}{dt}\right)_{first-order\ reaction} = -kVc \tag{2.28}$$

Where:

k Reaction's coefficient (T^{-1})

The foregoing terms can now be combined to yield the following mass balance for a completely mixed lake

$$V\ \frac{dc}{dt} = W(t) - Qc \pm kVc \tag{2.29}$$

The input in equation (2.29) is $W(t)$, the output is $c(t)$ and the system parameters are the constants, k, V and Q. The solution of Eq. (2.29) has two parts:

5. The complementary solution when there are no inputs of the concentration (i.e., $\mathrm{W(t)} = 0$).
6. The particular solution when $\mathrm{W(t)}$ has a specific form.

Thomann and Muller (1987) provide the solution to equation (2.29). The first part solution is

$$c = c_o exp\left[-\left(\frac{Q}{V} + k\right)t\right] \tag{2.30a}$$

$$c = c_o exp\left[-\left(\frac{1}{t_d} + k\right)t\right] \tag{2.30b}$$

where:

c_o Initial condition, at $t = 0$, of the concentration of the substance ($M\ L^{-3}$)
t_d Detention time of the lake (T)

This indicates that, as time increases, the effect of the initial condition c_o will gradually decay and disappear from the system. The time for the concentration to decrease to a given level depends on the reciprocal of the detention time of the lake, that is, the hydraulic flushing of the lake and the decay rate of the substance, k. Figure 2.39 shows this "flushing" of a lake initial condition (Thomann and Mueller, 1987).

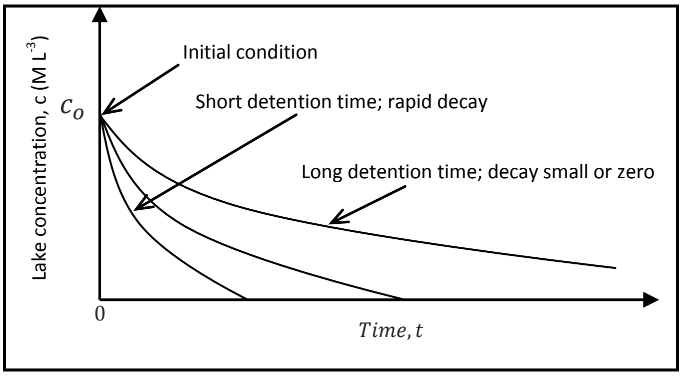

Figure 2.39: "Flushing" of lake condition, Equation (2.30), modified after Thomann and Mueller (1987).

2.3.5.3.2 One-dimensional models

One-dimensional water quality models for lakes and reservoirs can be divided into two types, longitudinal models and vertical models.

One-dimensional longitudinal models

One-dimensional longitudinal models are appropriate where vertical and lateral variations are negligible. The most likely application for this type of models is for *run-of-the-river* or *re-regulation* reservoirs where they may be more riverine rather than lacustrine at times.

Run-of-the-river reservoirs are shallow water bodies formed by a low-head dam or lock and dam. Re-regulation reservoirs are normally shallow reservoirs downstream of a deeper reservoir (Martin et al., 1999).

One-dimensional vertical models

One-dimensional vertically layered models for lakes and reservoirs are considered as the most useful water quality management tools. They have been particularly useful for simulating water temperature and water quality in deep reservoirs (Martin et al., 1999). Since layered models assume complete mixing in horizontal layers, horizontal momentum and lateral momentum are neglected and the vertical momentum equation is reduced to the hydrostatic approximation.

An important element of vertically layered models is vertical turbulent mixing. To quantify vertical mixing in lakes and reservoirs for one-dimensional layered models, two approaches are useful, *mixed-layer models* and *eddy-diffusivity models.* Both approaches have been widely used *(Martin et al., 1999).*

Mixed-layer models are used to compute the upper mixed-layer depth. These methods are generally based on an energy balance, where the kinetic energy due to forces such as wind and convection are compared with the potential energy contained in a layer of a stratified lake. If the energy available for mixing exceeds that required for complete mixing, the layer is mixed.

In eddy-diffusivity models, the vertical transport equation is used to determine the vertical exchange. The term of vertical turbulent diffusion is related to surface wind shear, the vertical dispersion coefficient is then reduced by a coefficient that relates the energy available for mixing to density differences that oppose mixing. Convective mixing is usually conceptualized in these models as either increasing the vertical dispersion coefficient or simply mixing the surface layer until a stable density profile is achieved.

Figure 2.40 shows the conceptual representation of a stratified reservoir (Orlob, 1984). The general one-dimensional mass conservation (advection-diffusion) equation for a one-dimensional vertical system is:

$$\frac{\partial C}{\partial t} = -w\frac{\partial C}{\partial z} + \frac{\partial}{\partial z}\left(E(z)\frac{\partial C}{\partial z}\right) \pm S \tag{2.31}$$

where:

t	Time (T)
C	Concentration of any constituent (M L^{-3}),
w	Vertical advection velocity (L T^{-1})
z	Vertical coordinate axis (+z is vertically upward)
$E(z)$	Coefficient of effective diffusion (L^2 T^{-1})
S	Sources or sinks of C(M L^{-3})

For a discrete volume like V_j in Figure 2.40 the mass conservation equation for substance C is given by (Orlob, 1983b):

$$\frac{\partial(V_j C_j)}{\partial t} = \underbrace{(Q_i C_i - Q_o C_o)_j}_{local\ advection} - \underbrace{(Q_z C)_j + (Q_z C)_{j+1}}_{vertical\ advection} + \underbrace{\left(E a_z \frac{\partial C}{\partial z}\right)_j - \left(E a_z \frac{\partial C}{\partial z}\right)_{j+1}}_{vertical\ diffusion}$$

$$+ \underbrace{C_j \frac{\partial V_j}{\partial t}}_{volume\ change} + \underbrace{V_j \frac{\partial C}{\partial t}}_{processes\ change\ (reactions)} \pm \underbrace{S}_{sources\ and\ sinks} \tag{2.32}$$

where:

t	Time (T)
C_j	Concentration of any constituent (M L^{-3})
z	Vertical coordinate axe (+z is vertically upward) (L)
V_j	Volume of the element j (L^3)
C_i	Input concentration of the constituent to element j (M L^{-3})
C_o	Output concentration of the constituent from element j(M L^{-3})
Q_i	Advective flow to element j (L^3 T^{-1})
Q_o	Withdrawal flow from element j (L^3 T^{-1})
E	Coefficient of vertical diffusion (L^2 T^{-1})
a_z	Horizontal area at depth a (L^2)
S	Sources or sinks of C(M L^{-3})

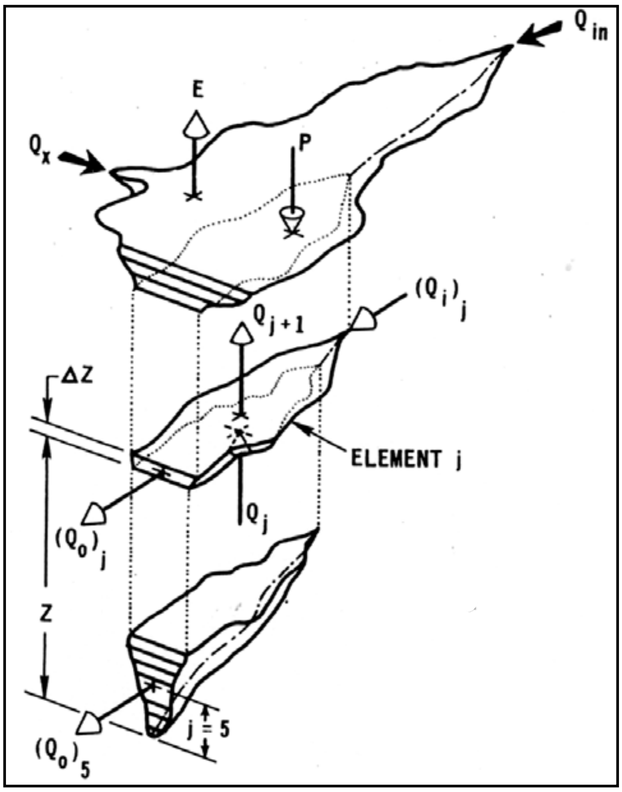

Figure 2.40: Conceptual representation of a stratified reservoir (Orlob, 1984). Q_{in}, inflow to reservoir; Q_x, local drainage; E, evaporation rate; P, precipitation rate.

2.3.5.3.2 Two-dimensional models

Two-dimensional water quality models for lakes and reservoirs can be divided into two types, laterally averaged models and depth averaged models.

Two-dimensional laterally averaged models

Laterally averaged two-dimensional models of lakes and reservoirs simulate water quality changes in the longitudinal and vertical directions. These models are generally applicable to longer, deeper, stratified reservoirs where lateral variations are negligible.

There are two types of these models; the “box” modeling approach and the full solution of the coupled hydrodynamic and water quality equations (Martin et al., 1999).

The box models solve the transport equation for each “box” accounting for advective flows and mixing exchanges occurring across each interface with any contiguous box. Mass exchange with the atmosphere and sediments are normally included. Sizes of boxes can be varied individually and boxes “stacked” to form one-, two-, or three-dimensional networks. The one-dimensional transport equation is solved in each direction. A good example of a box model is the U.S. Environmental Protection Agency’s Water Analysis Simulation Program model (WASP) (Ambrose et al., 1993). The WASP model can be linked with one-, two-, or three-dimensional hydrodynamic model.

The second approach is based on finite-difference solutions to the hydrodynamic equations coupled directly or indirectly with water quality algorithms. These are solutions to the conservation of momentum, water continuity and conservation of mass equations. For direct coupling, the transport equations are solved simultaneously with the hydrodynamic equations. The transport equations are typically solved using the same numerical grid and time step used for the solution of the hydrodynamic equations. Direct coupling of the hydrodynamic and transport simulations through the equation of state relating temperature and constituent concentrations to water density is necessary when temperature stratification or stratification by dissolved and suspended solids occurs, like in lakes and reservoirs. A good example of a box model is the U.S. Army Corps of Engineers' model CE-QUAL-W2 (Cole and Wells, 2008). For indirect coupling, results from the hydrodynamic simulation are saved in a file for later independent simulations of water quality (Martin et al., 1999).

Two-dimensional depth averaged models

Two-dimensional, depth-averaged models of circulation have also been applied to lakes and reservoirs. These models use the hydrostatic approximation and also solve the vertically averaged equations of continuity, x-momentum, y-momentum and constituent transport. These models are generally applicable to long, wide and relatively shallow water bodies or in cases where complete vertical mixing can be assumed (Martin et al., 1999).

Orlob (1984) and Watanabe et al. (1983) provide the theoretical formulation of two-dimensional depth averaged models, Reynolds equations, as follows:

- momentum equations

$$\rho\left(\frac{\partial u}{\partial t}+u\frac{\partial u}{\partial x}+v\frac{\partial u}{\partial y}\right)+\frac{\partial p}{\partial x}-\varepsilon_x\nabla^2 u-\rho\Omega v-F_{sx}+F_{bx}=0 \tag{2.33}$$

$$\rho\left(\frac{\partial v}{\partial t}+u\frac{\partial v}{\partial x}+v\frac{\partial v}{\partial y}\right)+\frac{\partial p}{\partial y}-\varepsilon_y\nabla^2 v-\rho\Omega u-F_{sy}+F_{by}=0 \tag{2.34}$$

- continuity equations

$$\frac{\partial u}{\partial x}+\frac{\partial v}{\partial y}=0 \tag{2.35}$$

where:

t	Time (T)
ρ	Density (M L^{-3})
u, v	Advection velocities along the principal axes, x and y (L T^{-1})
p	Pressure (M L^{-1} T^{-2})
ε	Turbulent exchange coefficient (M L^{2} T^{-1})
Ω	Coriolis parameter (T^{-1})
F_s	Surface friction (wind) force (M L T^{-2})
F_b	Bottom friction force (M L T^{-2})
∇^2	$=\partial^2/\partial x^2+\partial^2/\partial y^2$

The usual assumptions employed in adapting these equations to represent the behavior of a real system are (Orlob, 1984):

- The fluid is incompressible.

- All values of u, v and p are means; the fluctuating components are represented by turbulent exchange coefficients.
- Viscous and turbulent shear stresses are combined linearly.
- Vertical pressure distribution is hydrostatic.
- Variations in the Corilios parameter are negligible.
- The water column is fully mixed, that is, vertically homogeneous.

2.3.5.3.2 Three-dimensional models

Three-dimensional water quality models have not been used extensively in lakes and reservoirs, particularly for coupled hydrodynamic and water quality simulations. As is the case of the two-dimensional models, there have been two approaches used in three dimensional modeling: 1) box type modeling approaches using descriptive flow estimation and 2) models based on the solution of the hydrodynamic equations, conservation of momentum and continuity (Martin et al., 1999).

2.3.5.4 Numerical solution techniques

Water quality models are based upon the conservation of mass (or continuity since water is assumed to be incompressible) and conservation of momentum, which are expressed as a set of partial differential equations. In order to simultaneously solve these equations, a number of solution techniques have been developed. There are three main groups of numerical methods finite difference, finite volume and finite element (Figure 2.41).

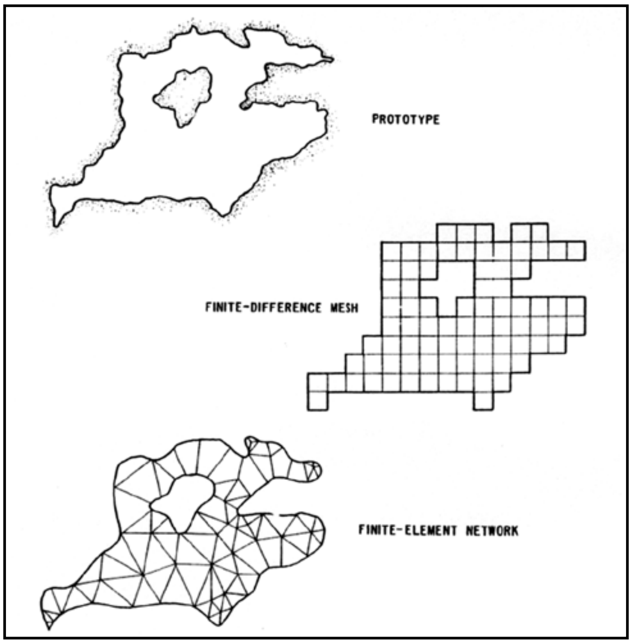

Figure 2.41: Conceptual representation of a shallow lake with a finite-difference mesh and a finite-element network (Orlob, 1984).

The finite difference method is built up from a series of nodes, the finite volume method is built up from a group of cells and the finite element method is made up of a group of elements, where each element is comprised of two or more grid points.

In the finite difference case, the differential equation is discretized over the numerical grid, and derivatives become difference equations that are functions of the surrounding cells. In the finite volume case, the fluxes through the cell network are tracked and the differential equations are integrated over the cell volume. For finite elements, a basis function is chosen to describe the variation of an unknown over the element and the coefficients of the bases functions are found by substituting the bases functions as solutions into the governing equations. Because finite difference methods are easier to implement and understand, these methods are more widely used (Socolofsky and Jirka, 2002).

A numerical method may further be explicit or implicit. An explicit scheme is the easiest to solve because the unknowns are written as functions of known quantities, which were computed at the previous time step and at upstream (known) locations. In an implicit scheme, the equations for the unknowns are functions of other unknown quantities, at the new time or on downstream locations not computed yet. In the implicit case, the equations represent a system of simultaneous equations that must be solved using matrix algebra. The advantage of an implicit scheme is that it generally has greater accuracy (Socolofsky and Jirka, 2002).

2.3.6 Modeling software tools (general-purpose software)

A model for a particular water management application can be constructed using either of the following sets of software tools:

1. Programming languages (to develop a new program)
- Traditional languages such as FORTRAN, BASIC, or C
- Object-oriented languages such as C++
2. Modeling environments or general-purpose software (to construct a model)
- Spreadsheets such as Lotus 1-2-3, Quattro Pro and Excel
- Object-oriented simulation packages such as STELLA and EXTEND
3. Generalized operational water quality models, using of an available model, such as CE-QUAL-W2.

Generalized operational models are existing models which have been already written and are widely used. *Generalized* means that the computer model is designed for application to a range of problems that deal with systems of various configurations and locations, rather than being developed to address a particular problem at a specific site. *Operational* means that the model is reasonably well documented and tested and is designed to be used by professional practitioners other than the original model developers. Generalized operational models should be convenient to obtain, understand and use and should work correctly, completely and efficiently.

The general-purpose computer programs may be used either in lieu of or in combination with the generalized operational models. These programs often serve as preprocessors to manipulate input data for the generalized operational models and/or as postprocessors to analyze, interpret, summarize, display and communicate simulation results.

The general-purpose computer programs have been reviewed by Wurbs (1995), Dahl and Wilson (2001) and Nirmalakhandan (2002).

2.3.6.1 Spreadsheet-based software

Numerous spreadsheet/graphics/database programs are on the market. *LOTUS 1-2-3*, *QUATTRO PRO* and *EXCELL* are among the more popular packages. In 1989, *LOTUS 1-2-3* has been used to model the fate of chemicals in an aquatic environment. In 1990, a spreadsheet model has been developed to simulate water quality in a chain of three lakes subject to pollution from runoff from urbanizing watersheds (Wurbs, 1995).

2.3.6.2 Dynamic simulation-based software

Commonly available dynamic simulation software are *EXTEND, STELLA or ITHINK, SIMULINK*, etc. Dynamic simulation packages typically feature a *flow diagram* interface, enabling modelers to assemble a flow diagram of the system being modeled using graphical icons. *STELLA* has been applied to a broad range of other problems in various disciplines, including water chemistry, biology and ecology (Wurbs, 1995).

2.3.6.3 Expert system software

An expert system is an advanced computer program that can, at a high level of competence, solve difficult problems requiring the use of expertise and experience. Expert systems may be stand-alone models or may be embedded as a component of a larger model. Two alternative types of software tools are available for developing expert systems: programming languages, such as *LISP and PROLOG* and shells. A shell is essentially an expert system with an empty knowledge base.

A knowledge-based system called *PUMP* was developed, using *PROLOG*, to guide reservoir operators in evaluating localized mixing alternatives for alleviating water quality problems associated with thermal stratification. The Cornell Mixing Zone Expert System (*CORMIX*), maintained by the Environmental Protection Agency Center for Exposure Assessment Modeling, is used for the analysis, prediction and design of aqueous toxic or conventional pollutant discharges into watercourses, with emphasis on the geometry and dilution characteristics of the initial mixing zone (Wurbs, 1995).

2.3.6.4 Equation solver-based software

Several types of equation solving packages with powerful mathematical capabilities have become available in the past two decades. Some of the more common ones are *MathCAD, Mathematica, MATLAB and TK! Solver (Nirmalakhandan, 2002)*. These software products provide capabilities for solving algebraic equations, performing differentiation and integration, matrix operations and statistical computations. It displays the results in numbers, tables, symbols and graphs.

2.3.6.5 Geographic information systems

A geographic information system *(GIS)* is a set of computer-based tools for storing, processing, combining, manipulating, analyzing and displaying data which are spatially referenced to the earth. There are some popular software packages such as: *ArcGIS, ARC/INFO and GRASS*. A number of studies of the impacts on water quality of human activities in watersheds have involved *GIS* applications. In implementing a program for water quality protection of supply sources, the Massachusetts Water Resources Authority used a *GIS* for data management and mapping in the delineation of ground-water and surface-water protection zones (Wurbs, 1995).

2.3.6.6 Computer-aided drafting and design

Computer-aided drafting and design (CADD) software is used for a variety of graphics applications in various fields, including water resources planning and management. CADD programs provide powerful drawing and specialized graphics capabilities. AutoCAD is the most popular of the various CADD packages. AutoCAD is a general-purpose two- and three-dimensional drafting system employed in a broad range of applications (Wurbs, 1995).

2.3.6.7 Graphics programs

Many of the water management models include built-in graphics. Graphics programs developed for specific water management modeling systems may often be used with other models as well. Numerous software products on the market focus specifically on creating graphs and charts such as: *Grapher, Sigma Plot and Surfer. Grapher* and *Surfer* are widely used in the water resources field (Wurbs, 1995).

2.3.7 Review of available, applicable hydrodynamic and water quality models for lakes and reservoirs

2.3.7.1 BATHTUB

BATHTUB from US Army Corps of Engineers Waterways Experiment Station, Environmental Laboratory (USACE/WES) performs steady-state water and nutrient balance calculations in a spatially segmented hydraulic network which accounts for advective and diffusive transport and nutrient sedimentation. Water quality parameters (total phosphorus, total nitrogen, chlorophyll-a, transparency and hypolimnetic oxygen depletion) are calculated with regressions from reservoir data (Dahl and Wilson, 2001).

2.3.7.2 TWQM

TWQM from the US Army Corps of Engineers Waterways Experiment Station, Environmental Laboratory (USACE/WES) computes the steady-state, longitudinal distribution of water quality downstream of a reservoir (Dahl and Wilson, 2001).

2.3.7.3 SELECT.

SELECT from the US Army Corps of Engineers Waterways Experiment Station, Environmental Laboratory (USACE/WES) predicts the vertical extent and distribution of withdrawal from a reservoir of known density and quality distribution for a given discharge from a specified location. Using this prediction for the withdrawal zone, SELECT computes the quality of the release for parameters (such as temperature, dissolved oxygen and iron) treated as conservative substances (Wurbs, 1995).

2.3.7.4 CE-QUAL-R1.

CE-QUAL-R1 from the US Army Corps of Engineers Waterways Experiment Station, Environmental Laboratory (USACE/WES) is a one-dimensional (vertical) reservoir water quality model. This model determines values for water quality parameters as a function of vertical location and time. A reservoir is conceptualized as a vertical sequence of horizontal layers with uniformly distributed thermal energy and materials in each layer (Wurbs, 1995).

2.3.7.5 BETTER

The Box Exchange Transport Temperature and Ecology of Reservoirs (**BETTER**) model incorporates a modeling approach in which the reservoir is segmented into an array of volume elements or boxes. The flow patterns of the reservoir are modeled as longitudinal and vertical flow transfers between the arrays of volume elements (Wurbs, 1995).

2.3.7.6 COORS and LARM

The Computation of Reservoir Stratification (**COORS**) model and the Laterally Averaged Reservoir Model (**LARM**) solve advection/diffusion equations in a vertical longitudinal plane through a reservoir. COORS and LARM both provide capabilities for predicting the temperature structure of deep reservoirs throughout the annual stratification cycle. The models also develop the temporal and spatial hydrodynamics of reservoirs to provide advective components for water quality models. CE-QUAL-W2 was developed by expanding LARM to include 20 water quality constituents (Wurbs, 1995).

2.3.7.7 EXAMS II

Exposure Analysis Modeling System (**EXAMS II**) from the U.S. Environmental Protection Agency (USACE/EPA) is an interactive modeling system that enables the user to specify and store the properties of chemicals and ecosystems. The solution can be steady-state or dynamic, and the model can be applied in one, two, or three dimensions (NG and Perera, 2005)**.** EXAMS combines chemical loadings, transport and transformation into a set of differential equations using the law of conservation of mass as an accounting principle (Ambrose et al., 1996)**.**

2.3.7.8 DYRESM

Dynamic Reservoir Simulation Model (**DYRESM**) is a one-dimensional hydrodynamics model for predicting the vertical distribution of temperature, salinity and density in lakes and reservoirs. DYRESM provides a mean of predicting seasonal and inter-annual variation in lakes and reservoirs, as well as sensitivity testing to long term changes in environmental factors or watershed properties. DYRESM can be run either in isolation, for hydrodynamic studies, or coupled to CAEDYM for investigations involving biological and chemical processes. DYRESMCAEDYM couples the one-dimensional hydrodynamics model DYRESM with the aquatic ecological model CAEDYM (Socolofsky and Jirka, 2002).

2.3.7.9 CAEDYM

The Computational Aquatic Ecosystem Dynamics Model **(CAEDYM)** is an aquatic ecological model that may be run independently or coupled with hydrodynamic models DYRESM or

ELCOM. CAEDYM consists of a series of mathematical equations representing the major biogeochemical processes influencing water quality. At its most basic, CAEDYM is a set of library subroutines that contain process descriptions for primary production, secondary production, nutrient and metal cycling, and oxygen dynamics and the movement of sediment (McKinney, 2004).

2.3.7.10 QUAL2K or QUAL2E

QUAL2K (or Q2K) from the U.S. Environmental Protection Agency (USACE/EPA) is a river and stream water quality model that is intended to represent a modernized version of the QUAL2E (or Q2E) model. The Enhanced Stream Water Quality Model QUAL2E is a one-dimensional (longitudinal) model for simulating well-mixed streams and lakes (McKinney, 2004). QUAL2E permits simulation of several conventional water-quality constituents in a branching stream system under steady conditions. The model uses a finite difference solution to the one-dimensional, longitudinal advective-dispersive mass transport and reaction equation (Ambrose et al., 1996). QUAL2E-UNCAS is an enhanced version of QUAL2E which provides capabilities for uncertainty analysis (Wurbs, 1995). QUAL2E divides a river system into computational elements consisting of headwaters, junctions and reaches. For each reach there are approximately 26 different input parameters that must be specified. Overall, the model requires roughly 100 separate input parameters (Hammond, 2004). QUAL2E is one of the most extensive water quality prediction models used. The configuration of the model is flexible, so that the user can change it for special applications. This model is so popular that it has been incorporated in many other models in one form or another (Palmer, 2001).

2.3.7.11 WQRRS

The Water Quality for River-Reservoir Systems (**WQRRS**) from the U.S. Army Corps of Engineers, Hydrologic Engineering Center (USACE/HEC) is a package of dynamic water quality and hydrodynamic models. This model is an extension of the CE-OUAL-W2 reservoir model that predicts the input quality and quantity for the river system tributary to the reservoir. The river portion of the model is similar to QUAL2. WORRS integrates the outputs from the river model into the reservoir model (Palmer, 2001). The WQRRS package includes the models SHP, WQRRSQ and WQRRSR, which interface with each other. The Stream Hydraulics Package (SHP) and Stream Water Quality (WQRRSQ) programs simulate flow and quality conditions for stream networks, which can include branching channels and islands. The Reservoir Water Quality (WQRRSR) program is a one-dimensional model used to evaluate the vertical stratification of physical, chemical and biological parameters in a reservoir. The WQRRSR and WQRRSQ programs provide capabilities for analyzing up to 18 constituents (Wurbs, 1995).

2.3.7.12 HEC-5Q

HEC-5Q from the U.S. Army Corps of Engineers, Hydrologic Engineering Center (USACE/HEC) simulates the operation of river-reservoir systems in basin-wide studies. Additional subroutines provide the water quality simulation module. The stream component is a one-dimensional unsteady model driven by a selection of available hydrologic routing models. The reservoir component is a one-dimensional (vertical) model. The model simulates DO, BOD, a selection of conservative and non-conservative constituents, and contains a phytoplankton option (Ambrose et al., 1996). The water quality module also includes an option for selecting the gate openings for reservoir selective withdrawal structures to meet user-specified water quality objectives at downstream control points (Wurbs, 1995).

2.3.7.13 TRISULA- DELWAQ

TRISULA- DELWAQ from Delft Hydraulics (Delft/Hydraulics) is a two- and three-dimensional hydrodynamic water quality model for lakes, estuaries and coastal regions. This model consists of equations that are solved numerically using finite element techniques. TRISULA is the hydrodynamic model and DELWAO is the water quality model. The water processes include all the major components of the primary productivity system as well as dissolved and detritus of the nutrients, spill and first-order decay processes. The models have been extensively used on large water resource projects throughout the world (Palmer, 2001).

2.3.7.14 CORMIX

Cornell Mixing Zone Expert System (**CORMIX**) from the U.S. Environmental Protection Agency (USACE/EPA) is a comprehensive system designed for environmental impact assessment of mixing zones resulting from wastewater discharge from point sources. CORMIX contains three major subsystems (EPA/CORMIX):

- **CORMIX1** is used to predict and analyze environmental impacts of submerged single port discharges to lakes, rivers and estuaries.
- **CORMIX2** may be used to predict plume characteristics of submerged multiport discharges.
- **CORMIX3** is used to analyze positively and neutrally buoyant surface discharges to lakes, rivers and estuaries with a high degree of accuracy.

2.3.7.15 WASP

The Water Quality Analysis Simulation Program (WASP) from the U.S. Environmental Protection Agency (USACE/EPA) is a generalized compartment modeling framework for simulating water quality and contaminant fate, and transport in surface waters, including rivers, reservoirs, estuaries and coastal waters. It is dynamic and can be applied in one, two, or three dimensions. Two major subcomponent models are provided with WASP5. The first one is the toxics subcomponent, TOXI5, which combines organic chemical process kinetics with simple sediment balance algorithms to predict dissolved and sorbed chemical concentrations in the sediment bed and overlying water column. While the second one is the dissolved oxygen/eutrophication subcomponent, EUTR05, which predicts DO, CBOD and phytoplankton dynamics as affected by nutrients and organic material. Transport can be driven by specified flows or by coupling to external hydrodynamic models (Ambrose et al., 1996). WASP6 comes with a data preprocessor that allows for the rapid development of input datasets, either by cut and paste or queried from a database. A Post-Processor provides an efficient method for reviewing model predictions and comparing them with field data for calibration. WASP6.1 is currently configured so that it can be linked to three hydrodynamic models. They include the Dynamic Estuary Model (DYNHYD5), River Hydrodynamics Model (RIVMOD-H) and Environmental Fluid Dynamics Code (EFDC)

(Hammond, 2004). WASP has been used for about twenty years and is a well-established water quality model (McKinney, 2004)**.**

2.3.7.16 CE-QUAL-W2

CE-QUAL-W2 from the US Army Corps of Engineers Waterways Experiment Station, Environmental Laboratory (USACE/WES) is a numerical, two-dimensional, laterally averaged model of hydrodynamics and water quality. This model is described in the next chapter. This model contains one module, which models both hydrodynamics and water quality. Since this software accounts for variations in longitudinal and vertical directions (not in lateral direction), this software is best used in situations where large variations in lateral velocities and water quality concentrations do not occur (NG and Perera, 2005)**.** CE-QUAL-W2 is based on an earlier reservoir model that has been enhanced by using the water quality processes of QUAL2. This has made this model a very powerful prediction model for reservoirs and reservoir flow management. The model has been extensively used in North America (Palmer, 2001).

2.3.8 Research deficits and demands for this study

As stated before, see chapter 1, it was addressed in the literature that there are no previous hydrodynamic and water quality models which had been developed for the selected Egyptian case study (Aswan High Dam reservoir or Lake Nasser), although this huge reservoir is very vital to Egypt; it provides more than 95 percent of the Egyptian freshwater budget. So, a hydrodynamic and water quality model for this reservoir needs to be developed based on a two-dimensional hydrodynamic and water quality code, CE-QUAL-W2. This code was selected for this research work according to the following reasons:

1. This code solves for gradients in the longitudinal and vertical directions but it assumes, that lateral gradients are negligible. Hence, it is most applicable to relatively long, deep reservoirs such as Lake Nasser reservoir which has a long, narrow shape, in particular of its southern part Lake Nubia. Lake Nubia is the selected case study for this research work.
2. CE-QUAL-W2 is a hydrodynamic **and** water quality model which means that there is no need to couple the water quality model with an external hydrodynamic model.
3. CE-QUAL-W2 is widely used to model lake stratification and water quality of lakes, reservoirs and rivers all over the world (more than 2400 applications) as stated in PSU (2008), see chapter 3. Moreover, this code is considered the preferred model of choice for reservoir water quality simulation (Fang et al., 2007).
4. It was applied for lakes and reservoirs in different climatic conditions, see chapter 3, such as arid and semi-arid climates, mostly in USA (Galloway and Green, 2003; 2007b; Nielsen, 2005; Sullivan et al., 2007), and subtropical climates, in Taiwan (Kuo et al., 2003; 2006; Kuo and Yang, 2000), China (Bittner, 2008; Busuiocescu and Meon, 1993; Plogmeier et al., 2010; Rühe et al., 2009) and Vietnam (Le and Meon, 2009; Lorenz and Le, 2010; Meon et al., 2009). The AHD (or Lake Nasser) reservoir is situated in a desert area; this area is in the transition zone between the subtropical climate and the Mediterranean climate (arid and semi-arid).

3. CE-QUAL-W2 characterization, study area and data collection

3.1 Model characterization

CE-QUAL-W2 is a two-dimensional, longitudinal/vertical, hydrodynamic **and** water quality model. The model has been applied to rivers, lakes, reservoirs, estuaries and combinations thereof (Cole and Wells, 2008). CE-QUAL-W2 is based on an earlier reservoir model that has been enhanced by using the water quality processes of QUAL2. This has made this model a very powerful prediction model for reservoirs and reservoir flow management (Palmer, 2001). CE-QUAL-W2 model has been widely used for reservoirs in the United States and around the world (refer to section 3.1.6), and is considered the preferred model of choice for reservoir water quality simulation (Fang et al., 2007). A full documentation of the model can be found in Cole and Wells (2008). A brief introduction to the model's history, capabilities, limitations, inputs and calibration methods follows.

3.1.1 CE-QUAL-W2 history

The original CE-QUAL-W2 model, known as LARM (**L**aterally **A**veraged **R**eservoir **M**odel), was developed by Edinger and Buchak (1975). This model simulated hydrodynamics, temperature and conservative water quality parameters such as TDS. Additional development allowed for simulation of multiple branches and estuaries and this model code was known as GLVHT (**G**eneralized **L**ongitudinal-**V**ertical **H**ydrodynamics and **T**ransport Model) (Cole and Wells, 2008). Water quality algorithms were later added and the resulting code was known as CE-QUAL-W2 Version 1.0 (Environmental and Hydraulic Laboratories, 1986). Subsequent releases included:

- Version 2.0 (Cole and Buchak, 1995)
- Version 3.1 (Cole and Wells, 2003)
- Version 3.2 (Cole and Wells, 2003)
- Version 3.5 (Cole and Wells, 2006)
- Version 3.6 (Cole and Wells, 2008)

Modifications to the model have included improvements to computational efficiency and accuracy, transport and mixing schemes as well as additional water quality algorithms, hydraulic structures and the ability to connect multiple waterbodies (Williams, 2007).

3.1.2 Capabilities and limitations

The CE-QUAL-W2 model is capable of predicting water surface elevations, velocities, temperatures and twenty-eight other water quality parameters such as dissolved oxygen, nutrients, etc. Additionally, over 60 derived variables including pH, TOC, etc. can be computed internally from the state variables and output for comparison to measured data (Cole and Wells, 2008). Geometrically complex waterbodies are represented by a finite difference computational grid. Multiple inflows and outflows to the waterbody are represented through point/nonpoint sources, branches, precipitation and other methods. Tools are available for modeling hydraulic structures such as spillways and pipes. Output from the model provides options for detailed and convenient analysis.

The model is limited by several assumptions and approximations used to simulate hydrodynamics, transport and water quality processes. The model solves for gradients in the longitudinal and vertical directions but it assumes lateral gradients are negligible. Hence, it is most applicable to relatively long, deep reservoirs as this assumption would be inappropriate for waterbodies with significant lateral variations. An algorithm for vertical momentum is not included. Table 3.1 shows the governing equations of the model assuming no channel slope. Results may be inaccurate in waterbodies with significant vertical acceleration. Water quality constituents represented by the model are complex interactions (Figure 3.1). The methods used to solve them are simplistic descriptions but have replicated observed processes in various systems with impressive results (refer to section 3.1.6). Several water quality processes are not simulated including sediment transport and accumulation, toxics, etc. (Cole and Wells, 2008).

Table 3.1: CE-QUAL-W2Governing equations without channel slope (Cole and Wells, 2008).

Equation	Governing equation assuming no channel slope and no momentum conservation at branch intersections
x-momentum	$\frac{\partial UB}{\partial t}+\frac{\partial UUB}{\partial x}+\frac{\partial WUB}{\partial z}=gB\frac{\partial \eta}{\partial x}-\frac{gB}{\rho}\int_{\eta}^{z}\frac{\partial \rho}{\partial x}dz+\frac{1}{\rho}\frac{\partial B\tau_{xx}}{\partial x}+\frac{1}{\rho}\frac{\partial B\tau_{xz}}{\partial z}$
z-momentum	$0=g-\frac{1}{\rho}\frac{\partial P}{\partial z}$
continuity	$\frac{\partial UB}{\partial x}+\frac{\partial WB}{\partial z}=qB$
state	$\rho=f(T_w,\Phi_{TDS},\Phi_{ss})$
Free surface	$B_\eta\frac{\partial \eta}{\partial t}=\frac{\partial}{\partial x}\int_{\eta}^{h}UBdz-\int_{\eta}^{h}qBdz$

3.1.3 Input data

CE-QUAL-W2 is a data intensive application. The data required for an application include bathymetric data, air temperature, dew point temperature, wind speed, wind direction, cloud cover, solar radiation, inflow and outflow volumes, inflow temperatures, precipitation, evaporation, water quality constituent concentrations and hydraulic and kinetic parameters. The availability and quality of these data directly affect model accuracy and limit usefulness (Williams, 2007).

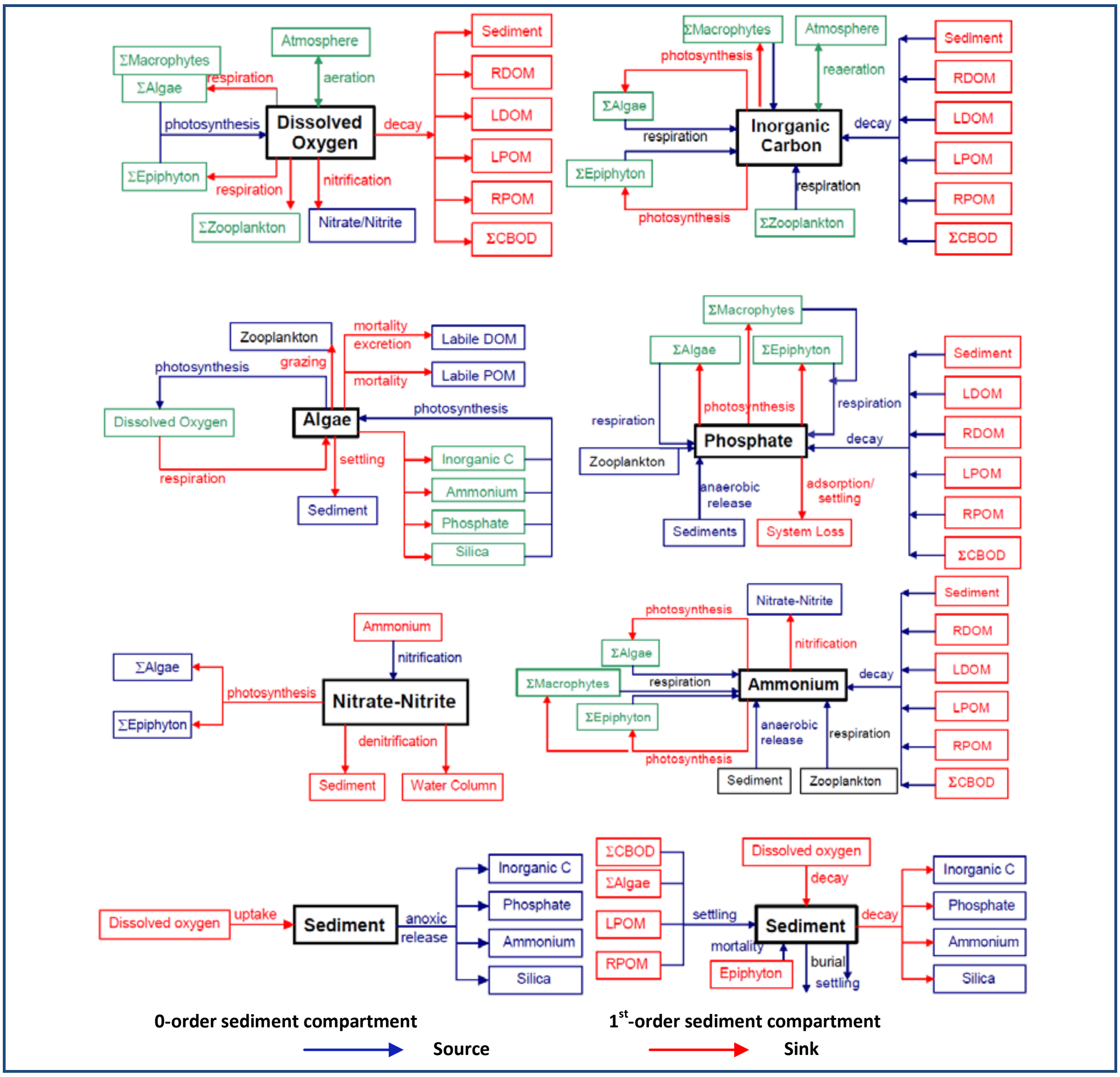

Figure 3.1: Internal flux between some compartments, each of them, and other compartments (Cole and Wells, 2008).

3.1.4 Computational grid

The computational grid, or the bathemetric data, is the term used for the finite difference representation of the waterbody. Grid geometry is determined by four parameters:

1. Longitudinal spacing (segment length).
2. Vertical spacing (layer height).
3. Average cross-sectional width.
4. Waterbody slope.

The longitudinal and vertical spacing may vary from segment to segment and layer to layer, but should vary gradually from one segment or layer to the next to minimize discretization errors (Cole and Wells, 2008). Figure 3.2 is a simple representation of a CE-QUAL-W2 grid. A waterbody consists of one or more branches. Tributary branches in a system are represented through additional branches which are linked to the main stem branch at a specific segment. One bathymetry file contains the dimensions from a single waterbody. A CE-QUAL-W2 model can include one or more waterbodies.

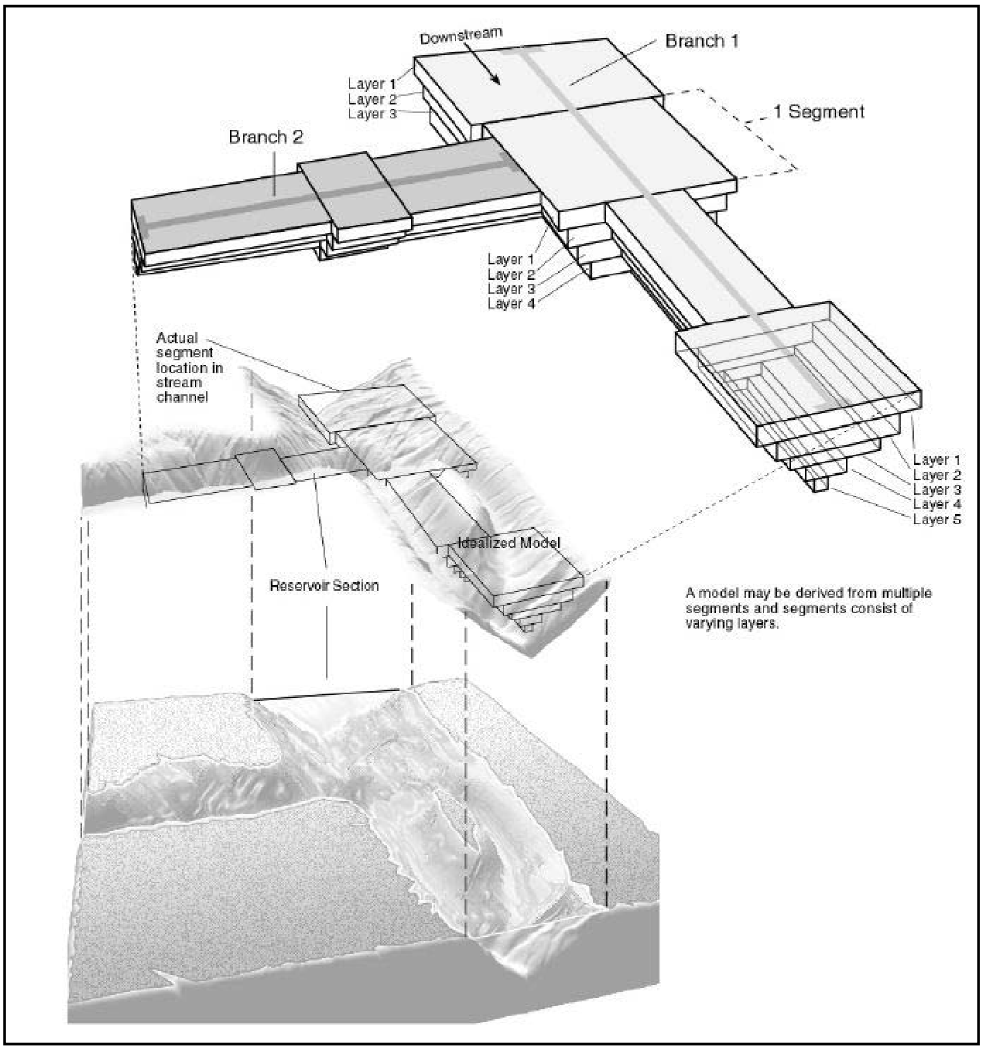

Figure 3.2: A simple representation of a CE-QUAL-W2 grid (Galloway and Green, 2003).

3.1.5 Calibration

Model calibration involves comparing observed, or measured, data to predicted, or simulated, results. Calibration is done iteratively until an acceptable fit of the predicted and observed data is achieved. There are no set guidelines for determining what an adequate fit is. The user must decide if and when the model is producing useful results. A typical sequence for model calibration is to begin with the water budget, followed by temperature and then water quality constituents (Cole and Wells, 2008).

Calibration statistics use two statistical propeties to compare simulated and measured in-reservoir observations, the absolute mean error (AME) and the root mean square error (RMS) (Cole and Tillman, 2001).

The absolute mean error (AME) indicates how far, on the average, simulated values are from measured values and is computed according to the following equation:

$$AME = \frac{\sum|Simulated\ value - Observed\ value|}{Number\ of\ Observations} \tag{3.1}$$

The root mean square error (RMS) indicates the spread of how far the computed values deviate from the observed data and is given by the following equation:

$$RMS = \sqrt{\frac{\sum(Simulated\ value - Observed\ value)^2}{Number\ of\ Observations}} \tag{3.2}$$

An AME of 0.5 °C means that the simulated temperatures are, on the average, within $\pm$ 0.5 °C of the observed temperatures, while an RMS error of 0.5 °C means that 67% of the simulated temperatures are within 0.5 °C of the observed temperatures (Cole and Tillman, 2001).

Other statistics may be used but the AME is preferred because it describes, on average, the difference between predicted and observed values (Cole and Wells, 2008).

Model calibration also requires testing model performance under different conditions than those used to generate the initial calibration model. This is called verification, or validation. In a reservoir application these may include sequences of differing inflow hydrology, reservoir elevations and release volumes (Cole and Wells, 2008).

3.1.6 Previous applications

CE-QUAL-W2 is widely used to model lake stratification and water quality of lakes, reservoirs and rivers all over the world. PSU website (2008) lists about 2400 of its applications worldwide. A brief summary to previous applications follows.

CE-QUAL-W2 was chosen to be applied to Cordell Hull Reservoir, Tennessee, USA. The application involved model adjustment and verification against observed data for April through October of two historical years. The verified model was then used to simulate hydrodynamics, temperature and DO within the lake for a wet, an average and a dry April through October for the purposes of (a) developing a better understanding of the interactions between stratification and hydrodynamics in this reservoir; (b) evaluating the water quality impacts of proposed submerged weir placement; and (c) evaluating the effects of hydropower operations on reservoir water quality characteristics (Howington, 1988).

CE-QUAL-W2 model and a zero-dimensional model (Vollenweider's type) were applied to Feitsui Reservoir, Taiwan, for water quality simulation and management. The simulation results indicated that the reservoir was nitrogen-rich but phosphorus turned out to be the limiting factor for alga growth. Reduction of phosphorus loading into the reservoir is recommended as a first step toward improving eutrophication problem. Both simplified and complex water quality models were shown to be an efficient tool to provide useful information for eutrophication management (Kuo and Yang, 2000).

CE-QUAL-W2 has been calibrated for temperature and algal/nutrient/DO interactions for Allatoona and West Point Reservoirs, Georgia, USA. When using the calibrated model as a management tool, one would have the most confidence using the model to investigate how operational changes would affect temperature and water quality – particularly DO. The model accurately captures the physics of both reservoirs. Any alteration in the physics should be predicted with a high degree of accuracy (Cole and Tillman, 2001).

CE-QUAL-W2 was used to simulate the water quality in the Feitsui Reservoir, Taiwan, in an effort to determine sound water quality management strategies. The model was calibrated and verified using data collected in 1996 and 1997. The calibrated model was used to simulate algal biomass (in terms of chlorophyll a) levels under various waste load reduction scenarios. Model results show that 50% reduction of the total phosphorus load will improve the existing water quality, shifting the trophic status from eutrophic/mesotrophic to oligotrophic (Kuo et al., 2003).

The U.S. Geological Survey, in cooperation with the Arkansas Game and Fish Commission, conducted a study to assess the impact of additional minimum flows, from 4.6 m^3/s (the existing minimum flow) to 22.6 m^3/s (the increased minimum flow), on the hydrodynamics, temperature and dissolved-oxygen concentrations in Bull Shoals Lake, Arkansas, USA, and on temperature and dissolved-oxygen concentrations in the downstream outflow. CE-QUAL-W2 was developed for Bull Shoals Lake, Arkansas. Scenarios included assessing the impact of (1) increased minimum flows and (2) increased minimum flows with increased water-surface elevation of 1.5 m greater than the initial water-surface elevation on outflow temperatures and dissolved-oxygen concentrations. With the increased minimum flow, water temperatures and dissolved oxygen concentrations increased in the outflow. Conversely, increased minimum flow with increased water-surface elevation decreased the outflow water temperature and dissolved-oxygen concentrations (Galloway and Green, 2003).

Because of its proven ability to accurately represent hydrodynamics and the ability to represent multiple algal groups, CE-QUAL-W2 was chosen to model Elephant Butte Reservoir, New Mexico, USA, to understand nutrient dynamics and algal response during a three year period (Nielsen, 2005).

CE-QUAL-W2 was applied to simulate the vertical-longitudinal hydrodynamics and water quality in Lake Erie during 1994. Central basin hypolimnetic anoxia was modeled to occur where the metalimnion intersected the sloping lake-bed, thus inhibiting diapycnal diffusion of dissolved oxygen (DO) through the water-column directly overlying the sediments. The study suggests that for 1994 conditions sediment oxygen demand, SOD < 0.2 gm^2d^{-1} would ensure a near bed DO concentration greater than 5 mg/L; sufficient to support fish life (Boegman, 2006).

The temperature distribution and movements of the density current of a deep warm monomictic reservoir (Lake Soyang, Korea), were simulated by using CE-QUAL-W2. The model could simulate temperature profiles with excellent agreement. Movement of the intermediate density current also was well simulated. The CE-QUAL-W2 model was useful in the prediction of temperature distribution and movement of density current in reservoirs, which implies merit for further employment of this model in water quality simulations (Kim and Kim, 2006).

Kuo, Lung, et al. (2006), have been done a study to quantify the mass transport, thermal stratification and water quality variations in the Te-Chi Reservoir (in the temperate climates) and

Tseng-Wen Reservoir (in the sub-tropical climates) in Taiwan through the application of the CEQUAL- W2 model. More specifically, the model results were used to quantify the cause-and-effect relationship between nutrient loads and water quality in these two reservoirs. Results of the model scenario runs reveal that a 30-55% reduction of the phosphorus loads will improve the water quality from a eutrophic/mesotrophic condition to oligotrophic condition in the Te-Chi Reservoir.

A CE-QUAL-W2 model was constructed to simulate hydrodynamics, water temperature, total dissolved solids and suspended sediment in Detroit Lake, Oregon, USA. The model was calibrated for calendar years 2002 and 2003 and for a period of storm runoff from December 1, 2005, to February 1, 2006. The calibrated model was used to estimate sediment deposition in the reservoir, examine the sources of suspended sediment exiting the reservoir and examine the effect of the reservoir on downstream water temperatures (Sullivan et al., 2007).

CE-QUAL-W2 was used for the evaluation of changes in input nutrient and sediment concentrations on the water quality of the Beaver Lake reservoir, Arkansas, USA, for the period of April 2001 to April 2003. Nutrients (nitrogen and phosphorus) and inorganic suspended solids concentrations were increased and decreased in the main tributaries to Beaver Lake to observe changes in the reservoir water quality from the different scenarios. Several conservative tracer scenarios also were used to observe the fate and transport in Beaver Lake and how the reservoir reacts to a simulated spill. These results can be used in the development of nutrient and turbidity criteria and standards for Beaver Lake. The methods also can be used as a prototype for assessing water-quality criteria in other reservoirs (Galloway and Green, 2007a).

Lake Powell has been simulated using CE-QUAL-W2. Previously the model was used at Lake Powell to simulate hydrodynamics, temperature and total dissolved solids with a reasonable degree of accuracy. An additional parameter, dissolved oxygen, has been added to the simulations and then calibrated with observed data to verify accuracy (Williams, 2007).

The U.S. Geological Survey, in cooperation with Colorado Springs Utilities and the Bureau of Reclamation, developed, calibrated and verified a hydrodynamic and water-quality model of Pueblo Reservoir, Colorado, USA, to describe the hydrologic, chemical and biological processes in Pueblo Reservoir that can be used to assess environmental effects in the reservoir. CE-QUAL-W2 modeling software, version 3.2 was used. The model was calibrated using data collected from October 1985 through September 1987 when measured water-quality data were available in the reservoir. The calibrated model was verified with data from October 1999 through September 2002 (WY 2000, 2001 and 2002). This 3-year contiguous period included various hydrologic conditions that allowed for verification of the model during a relatively wet year (WY 2000), an average year (WY 2001) and a dry year (WY 2002) (Galloway et al., 2008).

CE-QUAL-W2 was used in numerous projects by the Department of Hydrology, Water Management, and Water Protection, Leichtweiss-Institute, TU-Braunschweig, Germany. In 2001, the model was applied considering remote sensed data of the Möhnetalsperre Reservoir, Germany. The model was calibrated and validated and it produced satisfactorily the natural processes. The evaluation of the remote sensed data was carried out for the temperature of the water-atmosphere boundary layer and the mean algae concentration in the upper layers of the lake (Weihrauch, 2001). In 2008, CE-QUAL-W2 was developed and extended to study the effect of using macrophytes reed population in the littoral zone of Lake Chaohu in the Anhui Province, China, on the eutrophication and the blue green algae development. New strategies were developed in order to reduce the trophic status and to prevent the blue green algae from entering the water intake (Bittner, 2008; Bittner et al., 2008). In order to develop a decision support system (DSS) evaluation tool for the mitigation of eutrophication of Lake Chao (China), CE-QUAL-W2 was integrated with other two different models: Naxos-Paddy, which estimates the runoff from catchment, and Export Coefficient model, which estimates the nutrient loads from catchment (Plogmeier et al., 2010; Meon et al., 2009; Rühe et al., 2009). The developed DSS evaluation tool includes five scenario indicators for land use, population, point emission, waste water treatment plants and fertilizers.

3.2 Site characterization

Egypt lies in the northeast corner of Africa (Figure 3.3). The total area of Egypt is about one million Km^2. The Egyptian terrain consists of a vast desert plateau interrupted by the Nile Valley and Delta which occupy about 4% of the total country area. Most of the cultivated land is located close to the banks of the River Nile, its main branches and canals, are in the Nile Delta. The total cultivated area is about 3% of the total area of the country (FAO, 2005). Population is estimated at 73.4 million (2004) with an average annual growth rate of 1.8%, the population is expected to reach 95 million by 2025 (Attia, 2004). About 97% of all people are living in the Nile Valley and Delta (FAO, 2005).

The climate of Egypt is generally hot and dry, with mild winters. Rain falls during the winter season on the northern coasts and some parts of the Nile Delta (Figure 3.4). Rainfall is very low, irregular and unpredictable. The temperature varies between 8-18 ˚C in winter and between 21-36 ˚C in summer (Attia, 2004).

The Egyptian water resources system is composed of many interacting components. Fresh water resources include River Nile (about 55.5 BCM), groundwater (about 7.5 BCM) and precipitation (about 1.5 BCM). Egypt also practices the use of various types of marginal quality water, such as reuse of agricultural drainage water, reuse of treated domestic wastewater (about 1.0 BCM) and desalinated water. More than 80% of the total water demand is consumed by the agricultural sector. The per capita share of water is continuously declining; the present share is below 1000 CM/capita/year, and it might drop to 500 CM/capita/year in the year 2025, which would indicate “water scarcity”. In addition, there exists rapid degradation in surface and groundwater quality (Attia, 2004).

Figure 3.3: Map of Egypt (MWRI, 2005).

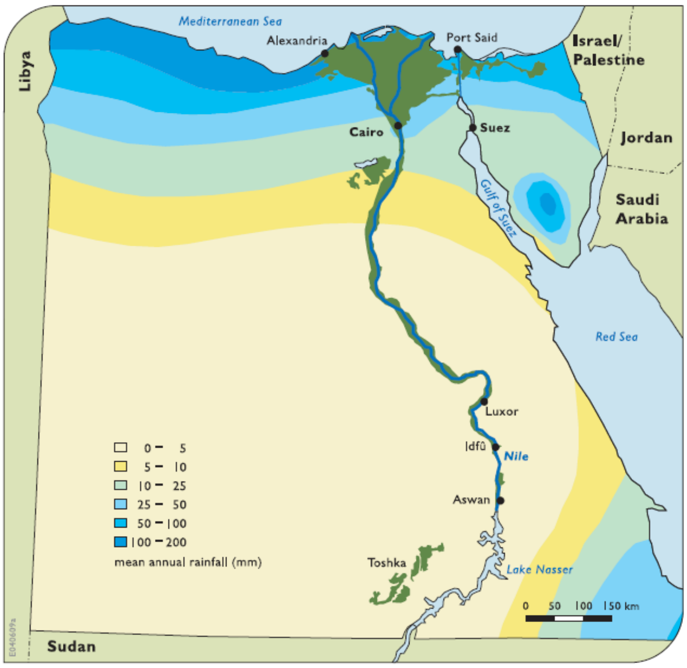

Figure 3.4: Average annual rainfall in Egypt (MWRI, 2005).

3.2.1 River Nile

Egypt receives about 98% of its fresh water from the River Nile. River Nile is either the longest or the second longest river in the world, Depending on how the measurement is done (Whittington and Guariso, 1983), it is about 6,700 km long. Its basin covers an area of about 3 million km^2 (one tenth of Africa), which is shared by ten countries as illustrated in Figure 3.5. The Nile basin stretches across 35 degrees of latitude and lies in three different climatic zones: tropical in the south, subtropical in the middle and arid desert in the northern section (Fahim, 1981; Park, 2001).

The Nile receives its water from two main sources: the equatorial East African Plateau, which feeds the White Nile, and the Ethiopian Highlands which feed the Sobat, the Atbara and the Blue Nile rivers (Fahim, 1981).

As illustrated in Figure 3.5, the White Nile begins from Lake Victoria, in fact from farther south at the Kagera River feeding the lake (Mohamed et al., 2005). The Victoria Nile flows about 90 km to Lake Kioga, and then on to Lake Albert. At Lake Albert the flow of the Kioga Nile merges with that of the Semliki River, which flows from Lake Edward and Lake George. From Lake Albert, the Albert Nile flows north about 168 km to Juba in the southern Sudan, and from there is termed the Bahr el Jebel, which spills into the Sudd, creating one of the world's greatest expanses of freshwater swampland (Fahim, 1981; Strzepeka et al., 2008; Whittington and Guariso, 1983). The Bahr el Jebel combines with the Bahr el Ghazel at Lake No, and numerous small tributaries, before leaving the Sudd region. In these Sudd Swamps, nearly half of the total flow is lost through evapotranspiration by wetland vegetation (Fahim, 1981; Strzepeka et al., 2008). Now it is termed as the White Nile, which flows about 964 km north to Khartoum, receiving the water of the Sobat River, whose source is in the Ethiopian Plateau, just south of Malakal. The Blue Nile originates in Ethiopia's Lake Tana and flows toward the main course of the White Nile in Northern Sudan. They meet at the city of Khartoum and from there the river has only one name, the Nile. Before the Nile leaves Sudan, a tributary known as the Atbara River joins the Nile's mainstream at a point located some 325 km north of Khartoum (Fahim, 1981). From Khartoum the Main Nile flows 3087 km through the extreme deserts of northern Sudan and Egypt to reach the Mediterranean. Twenty-four km north of Cairo, the Nile divides into two main branches: the Rosetta branch, flowing to the northwest, and the Damietta branch, flowing to the northeast (Whittington and Guariso, 1983). Figure 3.6 presents the longitudinal profiles of the Nile including the White Nile and Blue Nile headwaters.

The average annual natural inflow of the River Nile is about 84 BCM (MWRI, 2005). The Ethiopian Highlands provide about 86% of the annual inflow, while Equatorial lake sources contribute about 14%. The flood occurs during late summer and fall, arising from the spring rains in the Ethiopian mountains. During early summer when the flow at Aswan is lowest, the White Nile may contribute more than 75% of the total discharge. During the peak flood period in August, on the other hand, the White Nile may contribute only 6% of the discharge at Aswan (Whittington and Guariso, 1983). Figure 3.7 shows the relative monthly contributions of the White Nile and Blue Nile and the Atbara River to the discharge of the Main Nile throughout a mean year.

Further details on the Nile hydrology can be found in (Aboulhouda, 1993; Dumont, 2009; Fahim, 1981; Gupta, 2007; Said, 1993; Shahin, 2002; Sutcliffe and Parks, 1999; Whittington and Guariso, 1983).

Figure 3.5: River Nile and Nile basin, modified after MWRI (2005).

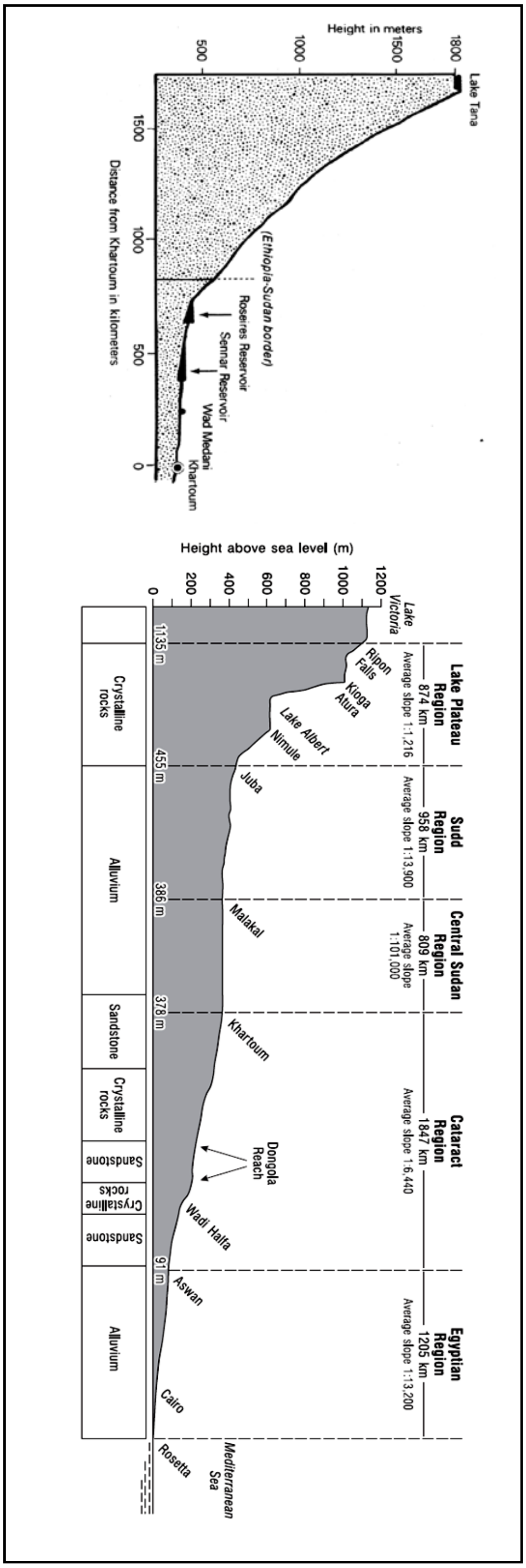

Figure 3.6: Longitudinal profiles of the Nile: (a) From the Blue Nile Headwaters to Khartoum city (Whittington and Guariso, 1983) (b). From the White Nile headwaters to the Mediterranean Sea (Woodward et al., 2007).

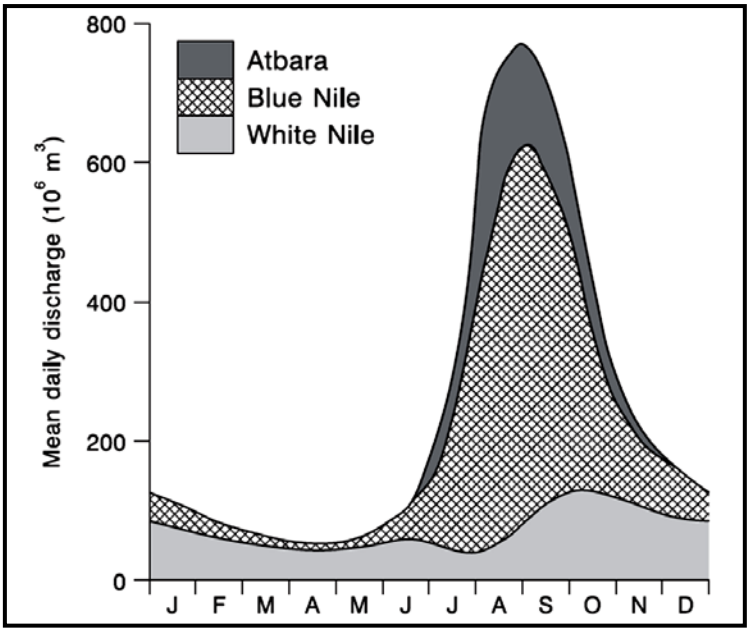

Figure 3.7: The relative monthly contributions of the White Nile and Blue Nile and the Atbara River to the discharge of the Main Nile throughout a mean year (Woodward et al., 2007).

Due to the seasonal nature of the Nile flow, several dams were built to control the Nile water for irrigation and hydropower generation. The first trial was in 2600 B.C. when ancient Egyptians built the world's oldest large dam Saad ElKafra (Garbrecht, 2006). In 2300 B.C., ancient Egyptians have used the Moeris Lake (Lake Qarun) at Fayoum, Egypt, as the first man-made flood regulator by digging a canal called the Bahr El Yussef carried the abundant water into this lake (Fahim, 1981). In the present time, there are eight dams which were built to control the river's flood for irrigation and hydropower purposes. The old Aswan Dam was built in 1902, but it was heightened twice afterwards to increase its storage capacity to 5000 million m^3 (Abdel Latif, 1984b). Aswan High Dam, the multi-objective project, is considered as the major regulator facility on the river (Attia, 2004).

3.2.2 Aswan High Dam (AHD)

For the first time in history, full control of the Nile water was achieved in 1970 after the construction of the Aswan High Dam (AHD) (Abu-Zeid and El-Shibini, 1997). The High Dam is located 6.5 km upstream of the old Aswan dam, about 950 km south of Cairo.

The first proposal to construct a huge dam at Aswan was proposed by Adrien Daninos, a Greco-Egyptian engineer, in 1948. In 1953 West Germany agreed to finance an engineering feasibility study of the project. The work was carried out for two years by two German firms, Hochtief and Dortmund. An international advisory panel was created and supported Hochtief and Dortmund's recommendations. Egypt contracted with Sir Alexander Gibb and Partners, a widely-respected British engineering firm, to review the work of both Hochtief and Dortmund, and the international advisory panel (Whittington and Guariso, 1983). The World Bank initially agreed to finance the huge project, but the Bank ultimately withdrew its support. Funds required to finance the project became available after the nationalization of the Suez Canal, and the Soviet

Union provided technical assistance (Strzepeka et al., 2008). In 1956, Soviet engineers reviewed the previous engineering feasibility studies, and, with some modifications, found them sound. Throughout the planning and construction phases the dam had the strong support of President Nasser. Just as Nehru subsequently claimed large dams are to be the temples of modern India, Nasser stated “In antiquity we built pyramids for the dead, now we will build new pyramids for the living” (Fahim, 1981; Scudder, 2003).

The construction of the dam took place in two steps. The first, 1960-1964, was the diversion of the river and the construction of coffer dams along its path to close off that part of its course where the main body of the dam was to be built (Said, 1993). The second step was the building of the main body of the dam. This step continued till 1968 and the hydroelectric power plant from 1967 to 1971 (Shahin, 2002). Fifty thousand engineers and workers took part in the implementation of this giant project. The Dam is a rock-filled structure provided with a grout curtain 213 m deep as shown in Figure 3.8 (Shahin, 2002). It is 3600 m long and pyramidal in shape (980 m wide at the base and 40 m wide at the top), and it rises 111 m above the Nile floor and 196 m above sea level. Water is released through six tunnels that are each of 14 m in diameter. These releases both regulate flow and produce electrical power from 12 generating units. A great reservoir was created in front of the High Aswan Dam (Strzepeka et al., 2008)**.** Full storage level (175 meters) of the reservoir was reached in 1975. The final cost of the main body of the High Dam and the power station was $820 million which were all paid off by 1978 (Said, 1993). A layout of the dam can be seen in Figure 3.9. Table 3.1 shows the technical data of the project. More details about the project can be found in (Fahim, 1981; Kinawy and Shenouda, 1993; Said, 1993; Whittington and Guariso, 1983).

As with other large projects, some side-effects have occurred, as anticipated, but the benefits far exceed these side-effects (Abu-Zeid and El-Shibini, 1997). The main aims of the construction of the High Aswan Dam can be summarized as follows (Abu-Zeid and El-Shibini, 1997):

- Full control of the Nile flow at Aswan in the far south of Egypt.
- Regulation of the discharge downstream of the dam to match the actual water needs for different requirements.
- Protection of the Nile Valley and Delta from high floods and drought hazards.
- Generation of cheap and clean hydroelectric power.
- Realization of horizontal land expansion by reclaiming new lands.
- Change in the system of basin irrigation (one crop per year), to perennial irrigation (two or more crops per year).
- Expansion in rice and sugar-cane cultivation to limit imports.
- Improvement of navigation through the Nile and navigable canals.

While the side effects can be summarized as follows (El-Shabrawy, 2009; Park, 2001):

- Siltation of AHD reservoir due to the capturing of fertile silt by the dam. This process affects the soil quality downstream.

- A growth of weeds at an epidemic scale in irrigation channels because of the inflow of silt-free water and the use of fertilizers in agriculture.
- Rising in groundwater levels due to the perennial irrigation and drainage.
- Significant effects occurred on the fisheries in the Nile and its coastal Lakes.
- Water losses due to evaporation and seepage of the reservoir water.
- Erosion of the Nile delta.
- Seismic stress due to the increased pressure of the large amount of reservoir water on the rocks.
- Displacement of people where about 100,000 Nubians were displaced in Egypt and Sudan when the reservoir filled.

Further discussions on the benefits and side effects of AHD can be found in (Abdel Latif, 1984b; Abu-Zeid and El-Shibini, 1997; Fahim, 1981; ICOLD, 1993; Saad, 2002; Said, 1993; Scudder, 2003).

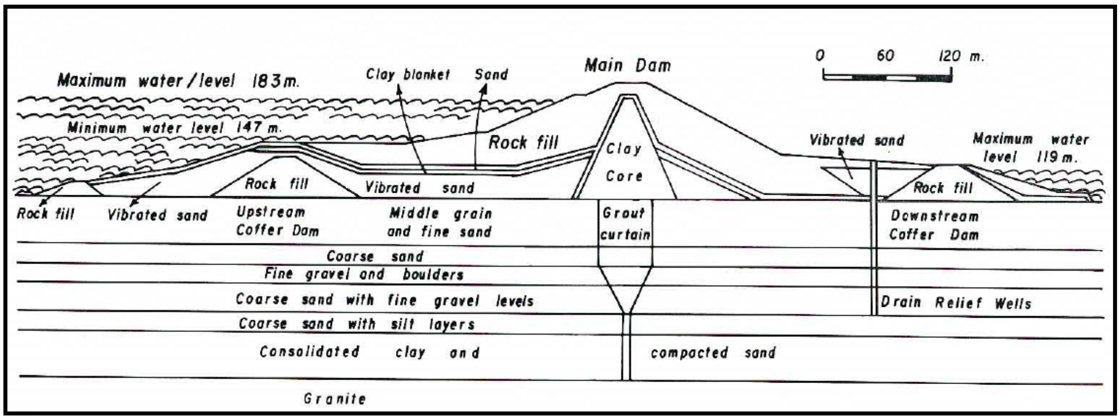

Figure 3.8: Cross section of the High Dam (Said, 1993).

Figure 3.9: Layout of Aswan High Dam (Google Maps)

Table 3.2: Aswan High Dam (AHD) technical data (Fahim, 1981).

Volume of materials of construction	42	Million m^3
Measurements		
Length at crest	3,820	m
Length of part in river	520	m
Length of right wing	2,325	m
Length of left wing	755	m
Maximum height above bed level	111	m
Width at crest	40	m
Width at base	980	m
River bed level	85	m
Crest level	196	m
Diversion channel		
Total length of diversion channel	1,950	m
Length of upstream open canal	1,150	m
Length of spillway tunnels and power house	315	m
Length of downstream open canal	485	m
Number of tunnels	6	
Inner diameter of tunnel	15	m
Maximum discharge into diversion channel	11,000	$m^3\ sec^{-1}$
The power station		
Total installed capacity	2.1	Million kw
Number of power units	12	
Capacity of each unit	175,000	kw
Design head	57.5	m
Total weight of hydropower equipment	30,000	tons

3.2.3 Aswan High Dam reservoir (or Lake Nasser reservoir)

AHD (or Lake Nasser) reservoir, one of the largest man-made lakes in the world, was formed as a result of the construction of AHD. The reservoir extends about 500 km upstream from the Aswan High Dam between the latitudes 23° 58' and 20° 27' N, and between longitudes 30° 35' and 33° 15' E (Figure 3.10). The current length of the submerged area is about 500 km, of which 350 km are within the Egyptian territory and is known as Lake Nasser. The 150 km stretch which lies in the northern part of Sudan is known as Lake Nubia. Generally, the reservoir is also known as Lake Nasser reservoir.

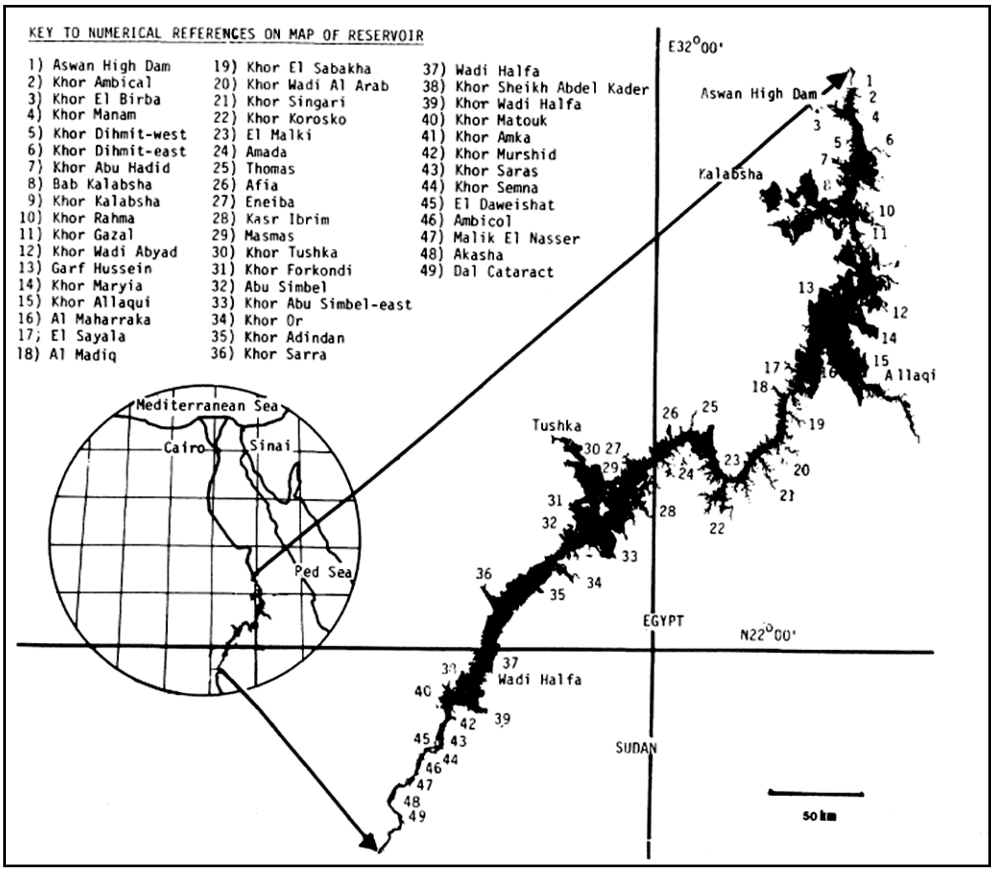

Figure 3.10: AHD reservoir (Whittington and Guariso, 1983).

AHD reservoir has a long, narrow shape with dendritic side arms or bays (Khours) extending in both the Egyptian and Sudanese stretches of the Nubian Nile. There are 100 important khors in Lakes Nasser and Nubia combined, 58 on the eastern and 42 on the western shores. The total length of the khor systems when the lake is full, at water level of 180 m AMSL, is nearly 3,000 km. The total surface area of the khors, the areas outside the main valley covered by water, is about 79% of the total lake surface. They contain about 55% of the total lake volume (El Shahat, 2000).

3.2.3.1 Reservoir climate

Lake Nasser reservoir is situated in a desert area, the climate is extremely arid. The area is in the transition zone between the tropical climate with summer rain and the Mediterranean climate with winter rain (Springuel and Ali, 2005). This area receives virtually no rainfall, except for occasional thunder storms in winter, roughly once every ten years (Abdel Latif, 1984b). The relative humidity is highest (40 - 41%) in December and January and lowest in the May and June (13 - 15%). The wind speed does not vary greatly all over the year, as the mean ranges from about 15 to 19 km h^{-1}. Its direction is mostly NW-NE (Abdel Latif, 1984b). Evaporation is very high, approximately 3,000 mm $year^{-1}$ (El Shahat, 2000).

3.2.3.2 Reservoir morphology

The reservoir has a maximum water depth of 130 meters (the mean depth is about 25 meters), or 182 meters above mean sea level (AMSL), and a total capacity of 162 BCM (about 15% of that for Lake Nubia). At this level the reservoir has a length close to 500 km (about 150 km for Lake Nubia) and an average width of 12 kilometers. The surface area of the reservoir at this maximum water level is 6540 km^2. Lake Nubia has about 14.8% of the total surface area. Figure 3.11 shows the topography of the reservoir region, while Figure 3.12 shows the water level – reservoir volume – surface area curve. Further details about the reservoir morphology can be found in (Abdel Latif, 1984a; b; Rashid, 1995; Said, 1993).

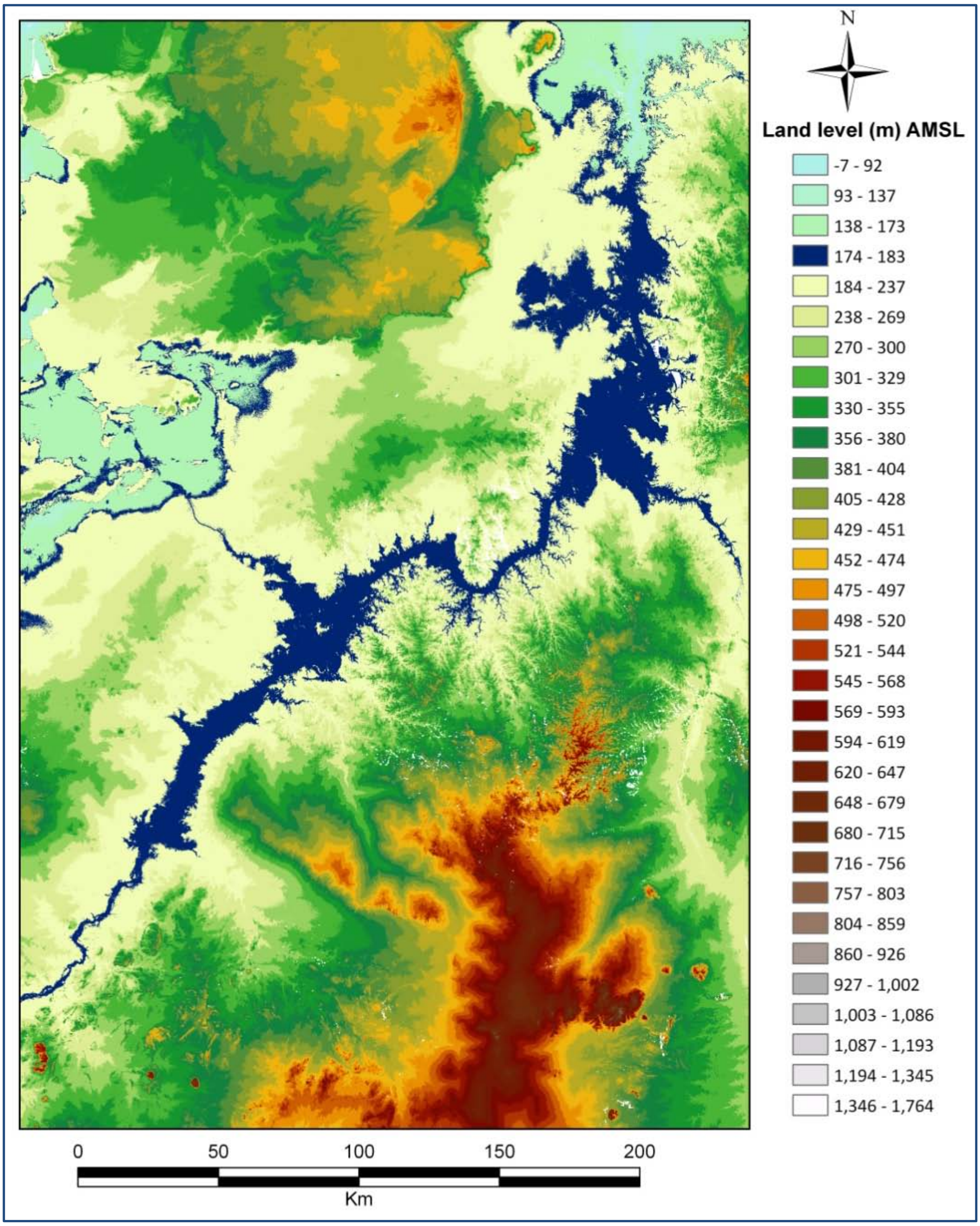

Figure 3.11: AHD reservoir region topography.

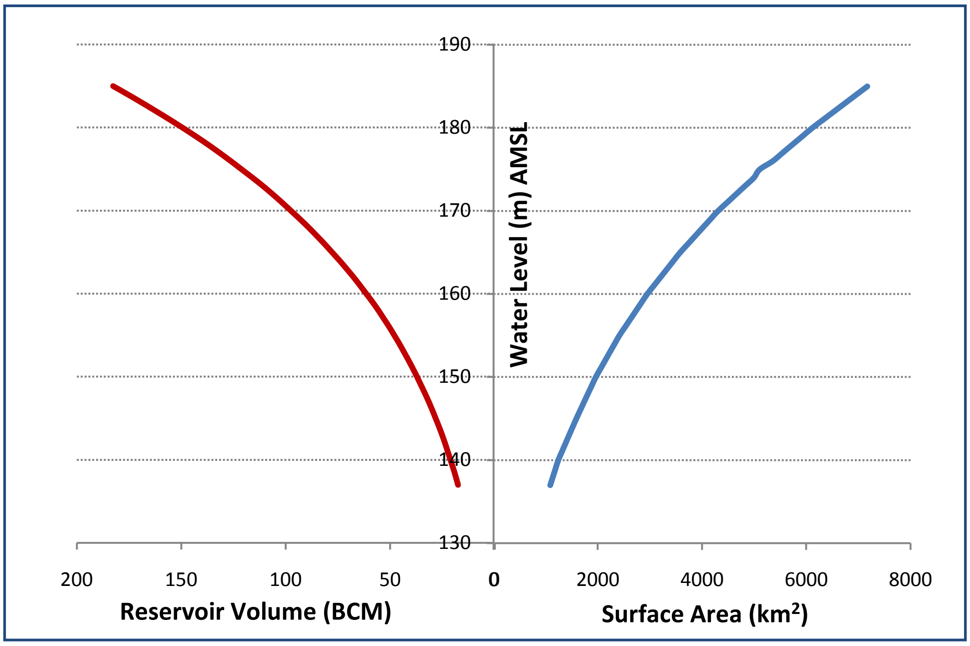

Figure 3.12: AHD reservoir water level – volume – surface area curve, the source of data Whittington and Guariso (1983).

3.2.3.3 Reservoir hydrology

The reservoir storage capacity is 162 km^3, divided into three zones as follows (Saad, 2000):

- 31 km^3 is the capacity for sediments deposited over 500 years. It lies between the bed and the 147 m water level.
- 90 km^3 is the live storage capacity, which lies between the 147 and 175 m water levels.
- 41 km^3 is the flood zone (emergency storage) for protection against high floods. It lies between the 175 and 182 m water levels.

AHD reservoir can be divided into three sections: riverine section (the southern sector), semi-riverine section (the middle sector) and lacustrine section (the northern sector) (Abdel Latif, 1984a). The riverine section, with all-year riverine characteristics, comprises the southern part of Lake Nubia, from the southern end to Daweishat (431 km upstream AHD). While the semi-riverine section, with riverine characteristics during the flood season (from the second half of July to November) and lacustrine characteristics during the rest of the year, extending from Daweishat (431 km upstream AHD) to Amada or Tushka (247 km upstream AHD). The lacustrine section, with all-year lacustrine characteristics, and extends from Amada or Tushka (247 km upstream AHD) to the High Dam.

The reservoir water levels change according to inflow and outflow rates. The highest water level was in November 2007 at 180.10 m AMSL, while the lowest was in 1988 at 151 m AMSL. Figure 3.13 shows the maximum and minimum reservoir water levels through the period from the year of 1966 to the year of 2005.

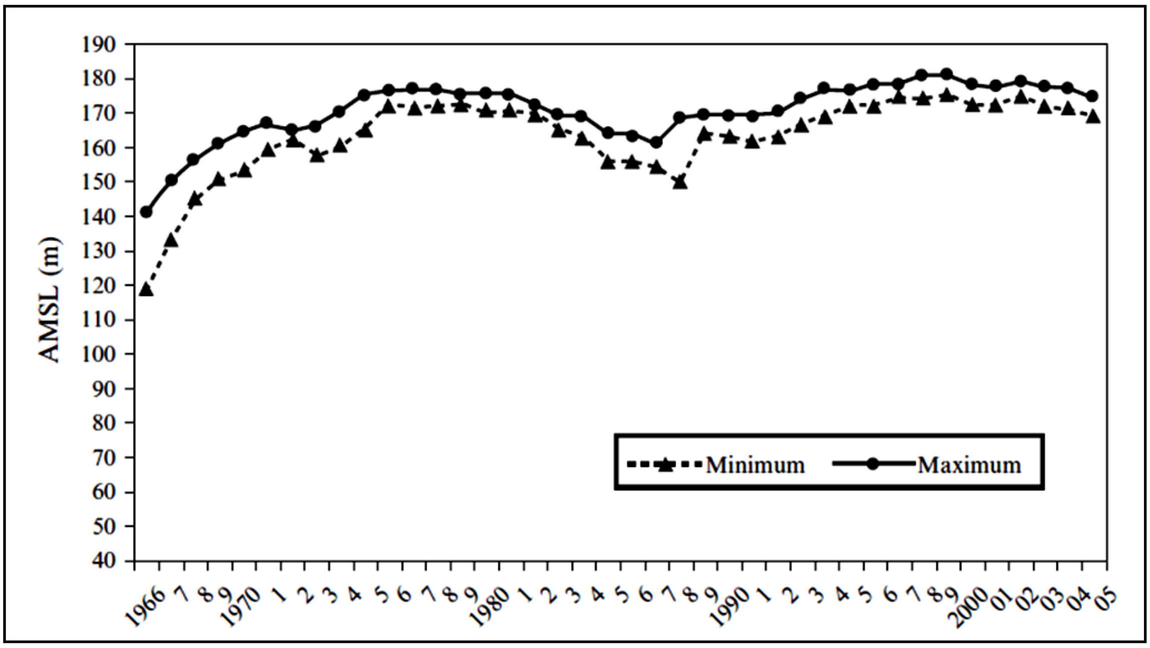

Figure 3.13: AHD reservoir water levels fluctuations through the period from 1966 to 2005 (El-Shabrawy, 2009).

The reservoir water velocity decreases as it approaches the High Dam. At the entrance of the reservoir, the flow velocity is about 0.5 m s^{-1}. This velocity is gradually reduced within a few kilometers to 0.1 - 0.2 m s^{-1} and in Lake Nasser to 0 - 0.03 m s^{-1} (El Shahat, 2000).

The Nile flood usually carries about 134 million tons of suspended matter (silt) per year, based on the average yearly inflow of 84 km^3 (Saad, 2000). It is heavily loaded with inorganic clay, silt and sand, and organic debris (detritus) (Rashid, 1995). About 98% of the annual sediment load occurred during the flood season (Abdel Latif, 1984b). Almost all the silt brought by the River Nile is deposited within Lake Nubia, most of it being deposited south of Halfa, where a new delta is in formation, and only the fine silt enters Lake Nasser (Springuel and Ali, 2005). Deposition in the reservoir is governed by a number of factors; the major factor is the sudden decrease of flow velocity of the flow as soon as it reaches the open area of Lake Nubia. The maximum silt thickness, the area of most intense deposition, recorded in Lake Nubia until 1998 was at Kagnarty (Station No. 6, 394 km upstream AHD) where this layer was about 60.2 m (El-Manadely et al., 2002).

The water losses of the reservoir are mainly due to evaporation and seepage. Annual evaporation losses are estimated at about 9.6 km^3, while annual seepage losses are estimated at 0.05 km^3 (Saad, 2000).

3.2.3.4 Reservoir water quality

The Egyptian Ministry of Water Resources and Irrigation, MWRI, is carrying an environmental monitoring program for AHD reservoir water quality (MWRI, 2002). MWRI realized the importance of protecting the reservoir from pollution, as it is almost the sole source of freshwater to the country. MWRI suggests that AHD reservoir and a strip of 20 km wide on both sides should be announced as a natural protectorate.

Generally, from samples collected before and after the flood, the reservoir water has good physical and chemical characteristics for use (Saad, 2000). The thermal pattern of the reservoir is warm monomictic; the reservoir stratifies in summer, and mixing occurs in winter (Williams, 2005). Transparency is affected by the turbidity caused by silt and clay of riverine origin. It is particularly strong in the flood season (Figure 3.14).

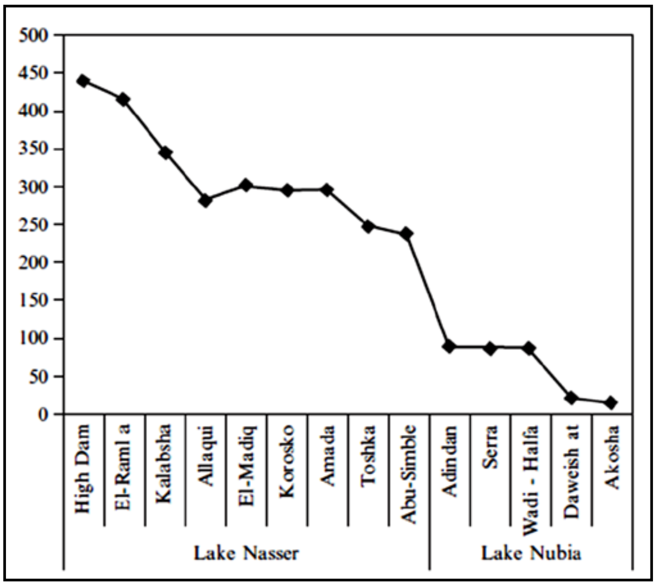

Figure 3.14: AHD reservoir Secchi disk transparency (cm) at some stations (El-Shabrawy, 2009).

Water temperature ranges between 15.0 °C in February and 32.4 °C in August, pH is normally alkaline and the reservoir trophic status is eutrophic (El Shahat, 2000). Dissolved oxygen is usually high during November-April and low during May- October. In surface waters, concentrations often vary between 3 and 12mg l-1, but in deeper layers, they may be much less: 0 – 8 mg l-1. Salinity is low at all times and generally <200 mg/L. Recorded concentrations for ortho-phosphate have ranged from 0.001 to 2.2mg/L (Williams, 2005).

The phytoplankton community is composed of blue-green algae, diatoms, green algae and dinoflagellates. Blue-greens dominate the community during spring and summer, while diatoms dominate the community only in winter (Habib, 2000).

Further details about the reservoir water quality can be found in (Abdel Latif, 1984a; Craig, 2000; El-Shabrawy, 2009; El Moattassem et al., 1993).

3.3 Data collection

The southern part of Lake Nasser (AHD) reservoir which lies in Sudan, Lake Nubia, was chosen as the case study of this work due to the availability of different required data to generate the input files of the two dimensional hydrodynamic and water quality model CE-QUAL-W2. Two different data sets were collected during low flood period for two different years; January 2006 (7.-19.) and February 2007 (2.-14). Each collected data set includes:

1. Morphological data, which include the bathymetry of the reservoir.
2. Hydrological data, which cover the inflows and water levels.

3. Meteorological data.
4. Hydrodynamic and water quality data, which include physical, chemical and biological measurements for the in-reservoir water.

3.3.1 Morphological data:

The bathymetric data of Lake Nubia for January 2006 and February 2007 were provided by Nile Research Institute (NRI) and Water Resource Research Institute (WRRI), National Water Research Center (NWRC), Egypt. Figure 3.15 shows Lake Nubia bathymetry of January 2006.

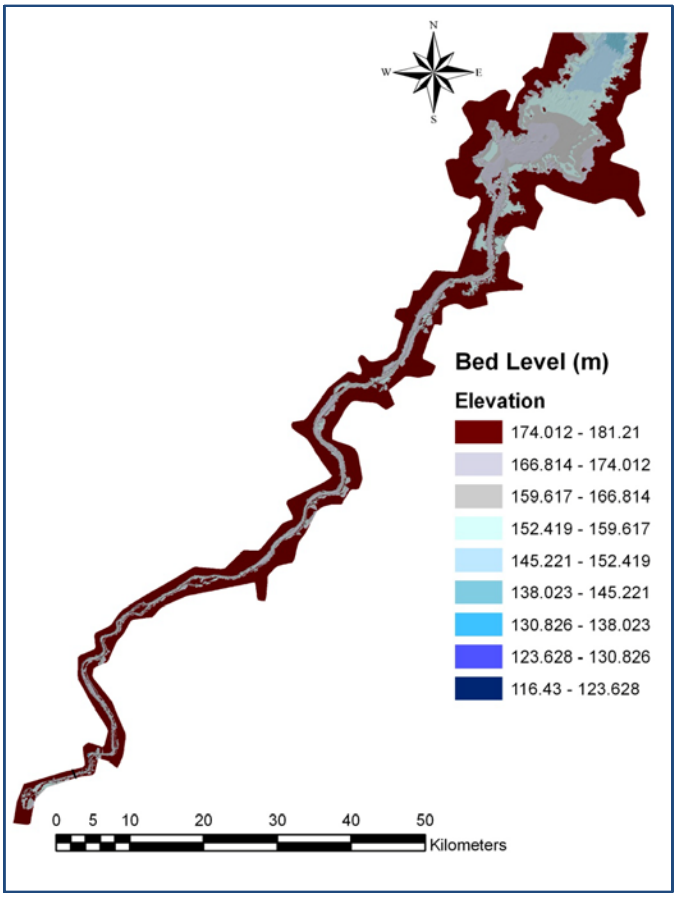

Figure 3.15: Lake Nubia bathymetry, January 2006.

3.3.2 Hydrological data:

The hydrological data of Lake Nubia for January 2006 and February 2007 were provided by Nile Research Institute (NRI), National Water Research Center (NWRC), Egypt. The hydrological data include Lake Nubia inflows and water levels.

Lake Nubia has only one inflow source, River Nile, at its upstream end. The inflow data were specified as daily average values from one gauging station at the upstream end of the reservoir. Figure 3.16 shows Lake Nubia inflows, monthly average, through the period from August 2005 to July 2007.

Lake Nubia water levels data and the bathymetric data sets were used to generate a Triangulated Irregular Network (TIN) of Lake Nubia water depths, by using Geographic Information System (GIS) software (Figure 3.17).

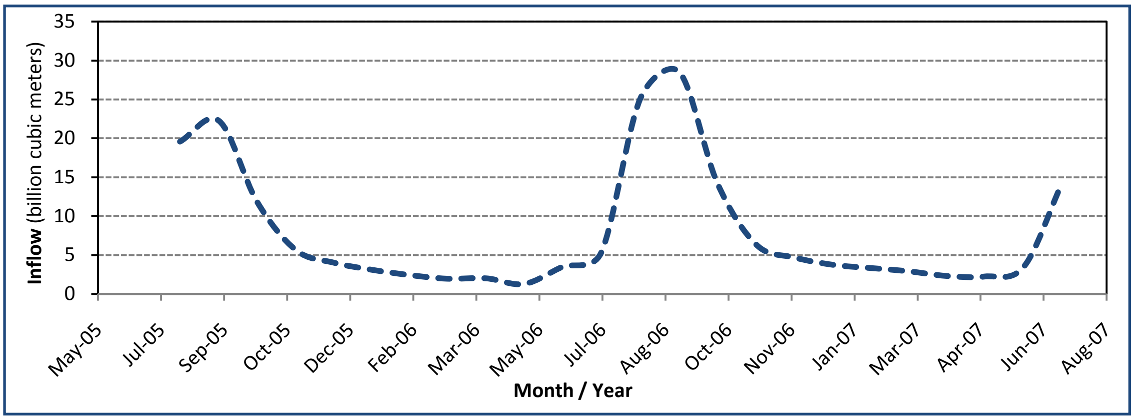

Figure 3.16: Lake Nubia monthly inflows (2005 – 2007)

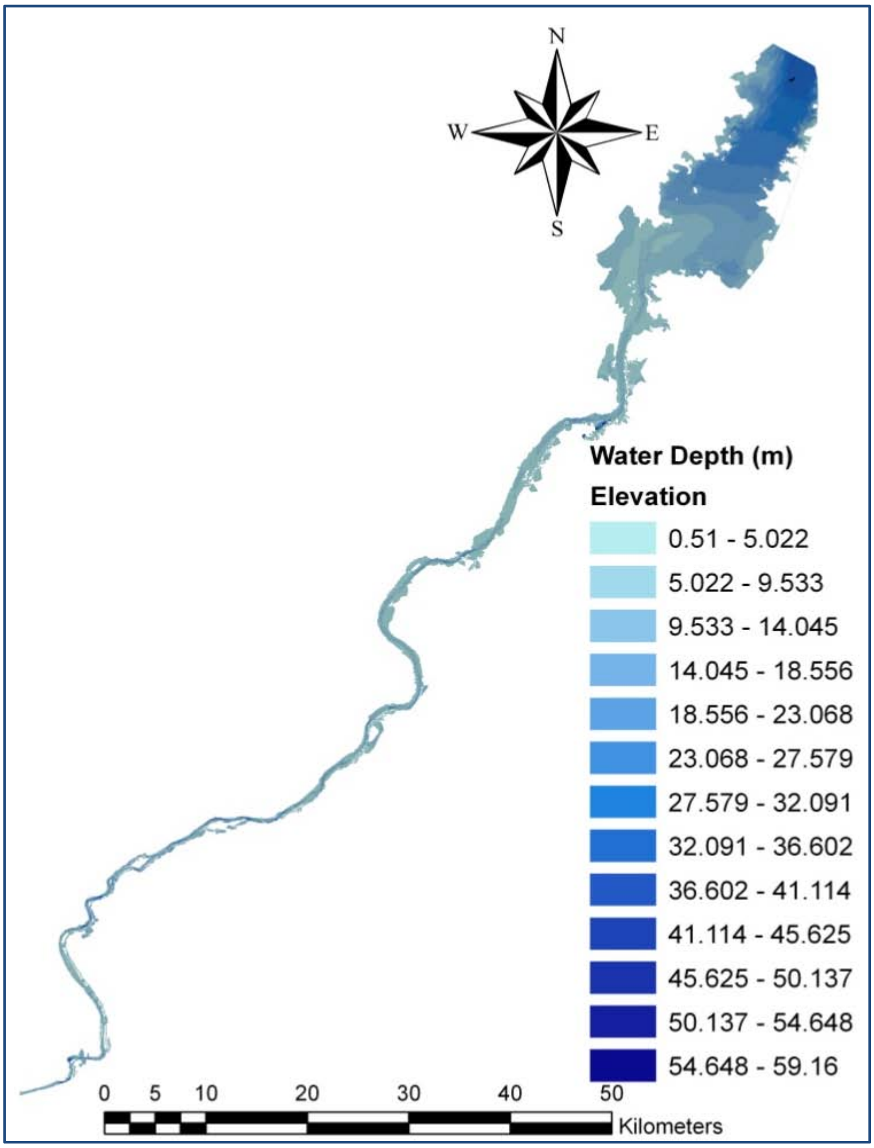

Figure 3.17: Lake Nubia water depths, January 2006.

3.3.3 Meteorological data:

The meteorological data were obtained for the nearby local meteorological station, Wadi Halfa station, from the internet (website of (Weather Underground)). The recorded data were given for every six hours. The meteorological data cover air temperature, dew point temperature, wind speed and direction, and cloud cover. Figures 3.18 and 3.19 show air temperature and dew point temperature and wind speed, respectively, for January 2006.

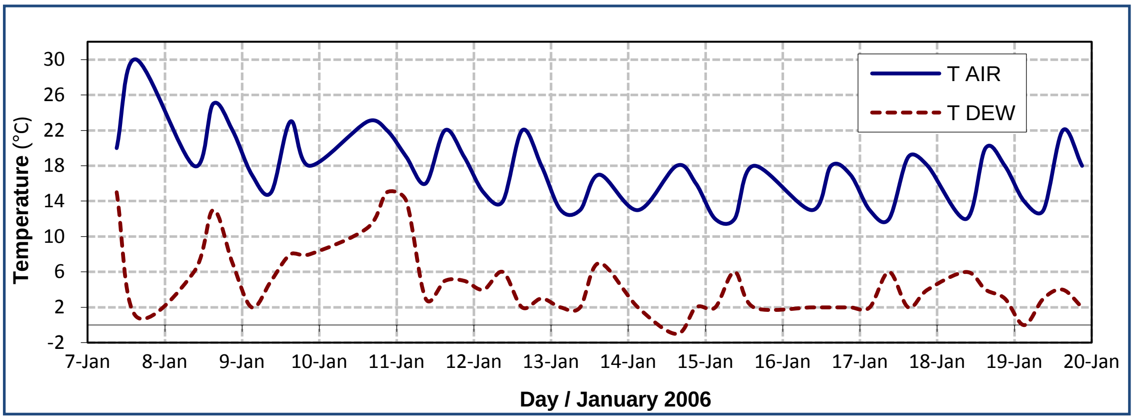

Figure 3.18: Lake Nubia air temperature and dew point, January 2006.

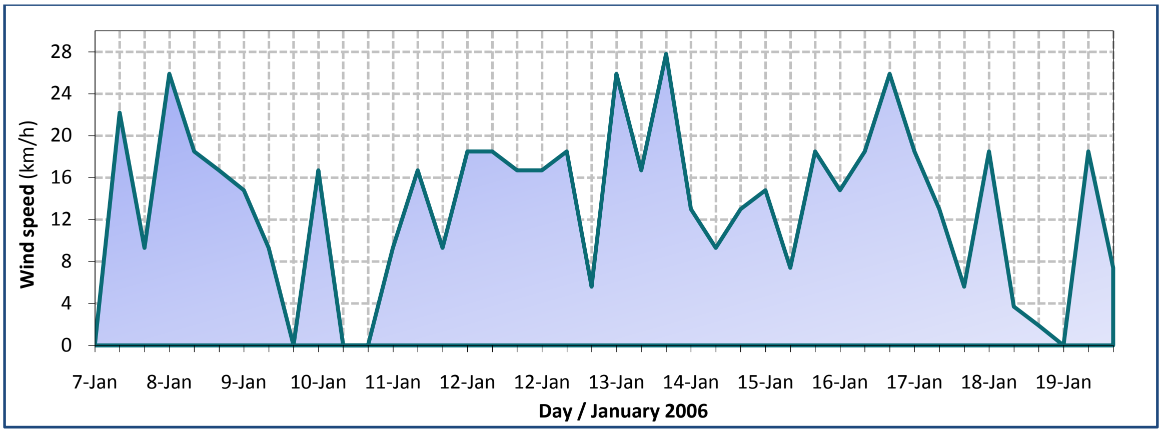

Figure 3.19: Lake Nubia wind speed, January 2006.

3.3.4 Hydrodynamic and water quality data

Hydrodynamic and water quality data of Lake Nubia for January 2006 and February 2007 were provided by Nile Research Institute (NRI), National Water Research Center (NWRC), Egypt. The measured hydrodynamic and water quality data consist of water temperature, dissolved oxygen, chlorophyll-a, orthophosphate, total phosphorus, nitrate-nitrite, ammonium, total dissolved solids, total suspended solids, turbidity, fecal coliform and pH. In site parameters have been analyzed in a mobile Water Quality Laboratory, while other parameters have been analyzed in the NRI Water Quality Laboratory. The collected samples were analyzed using standard methods of American Public Health Association (APHA) (Clescerl et al., 1999).

In-reservoir temperature and constituent concentration profiles were measured at 18 sampling stations positioned along the longitudinal axis of Lake Nubia, as seen in Figure 3.20. At each station, the water samples were collected from the surface and at 25, 50, 65 and 80% of depth at three different vertical axes (east, middle and west). Chlorophyll-a samples were collected from different two zones, Lighted and dark zones. Fecal coliform samples were collected from the surface and at 50 and 80% of depth. Figure 3.21 shows surface measured concentrations of some selected parameters at different sampling stations along Lake Nubia. Appendix 1 includes sample measured hydrodynamic and water quality records of Lake Nubia.

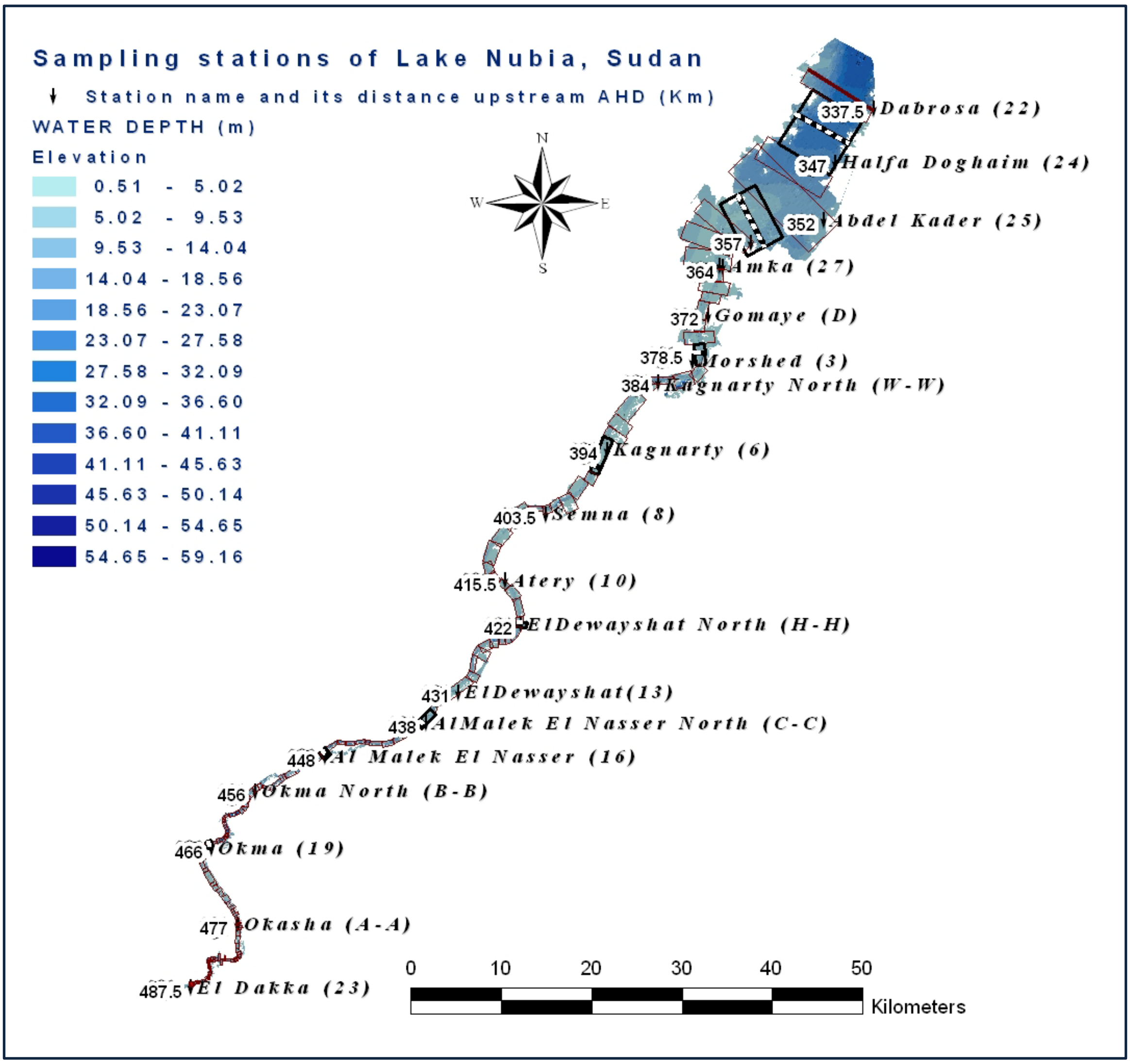

Figure 3.20: Lake Nubia sampling stations.

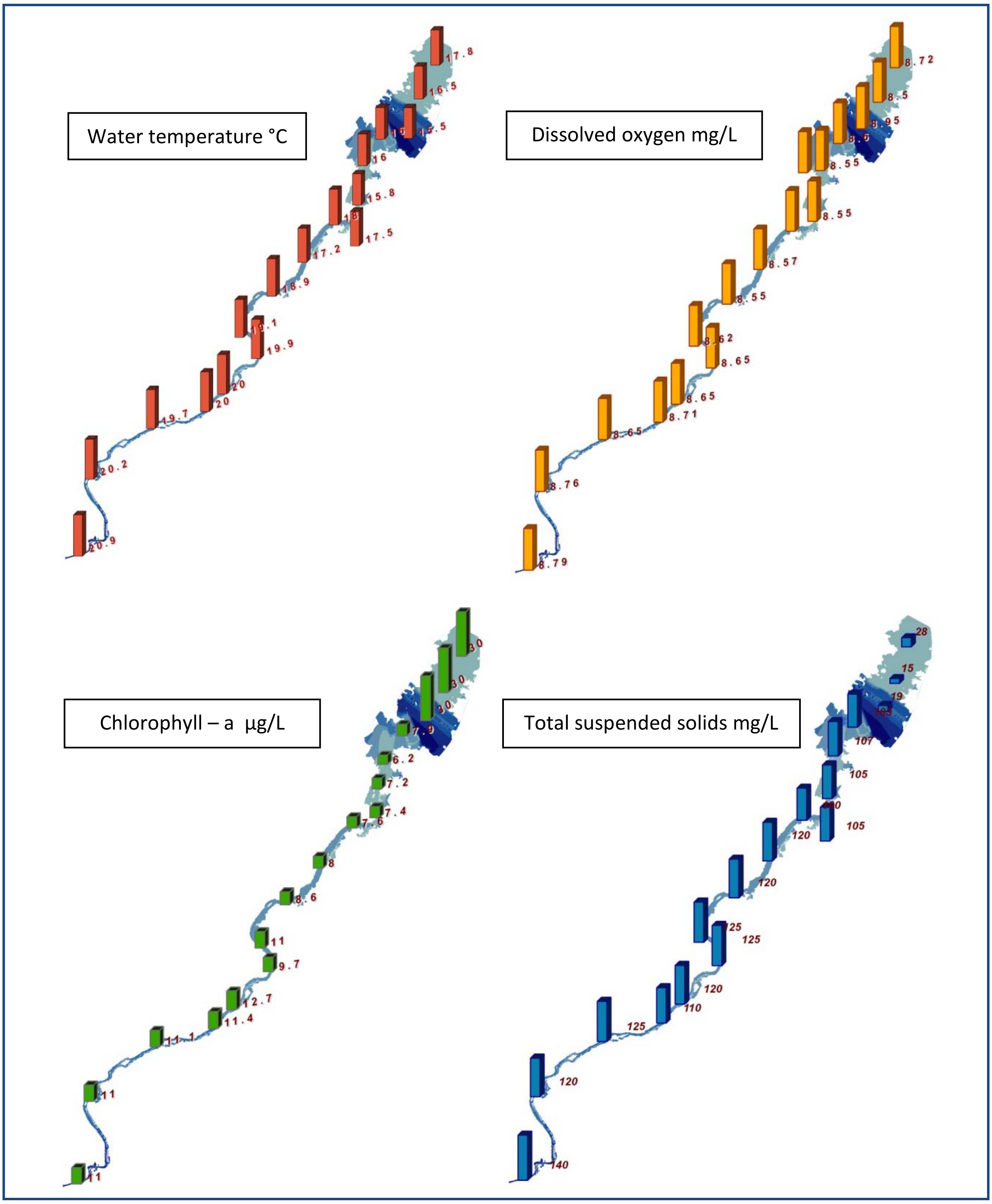

Figure 3.21: Surface measured concentrations of some selected parameters at different sampling stations along Lake Nubia, January 2006.

4. Model development

In this chapter, the description of the CE-QUAL-W2 model development for the case study, Lake Nubia, is presented. The CE-QUAL-W2 code description, the case study characteristics and the collecting of the model development required data are discussed in the former chapter. The following sections of this chapter discuss the various model input files including bathymetry (section 4.1), initial conditions (section 4.2), meteorological data (section 4.3), boundary conditions (section 4.4) and the associated required assumptions (section 4.5). The calibration and verification processes of the model are discussed in section 4.6. Section 4.7 discusses the calibration and verification results of the model. Conclusions and recommendations for future works are presented in sections 4.8 and 4.9, respectively.

4.1 Bathymetry

The development of the Lake Nubia CE-QUAL-W2 model begins by creating the bathymetry. Bathymetry includes the depth, length, width and orientation of each cell used in a grid to describe the reservoir. Getting a high-quality bathymetry of the reservoir is vital to obtain accurate model results. Ideally, a three-dimensional digital elevation set would be used, but unfortunately, this information was not available. Instead, a Triangular Irregular Network (TIN) built from cross sectional data of the reservoir was used. Figures 4.1 and 4.2 present Lake Nubia model bathymetry with plan and longitudinal section views of the grid, respectively. Cross section view of some bathymetry segments can be seen in Figure 4.3.

The Lake Nubia CE-QUAL-W2 model bathymetry was modeled by establishing a finite difference grid, using ArcGis (version 9.2) software, consisting of three main branches with 202 segments along the longitudinal axis and 27 vertical layers of 2 m depth. The average segment length is about 750 m long. A specific width was assigned to each cell of the model grid. Ten segments along the reservoir were chosen to be the control stations (Figure 4.1 and Table 4.1) which are used to represent the model results.

Table 4.1: The control stations characteristics of Lake Nubia.

Sampling site	Segment number	Site name	Distance upstream AHD (km)
St. 1	2	El-Daka (23)	487.5
St. 2	100	Okma (19)	466
St. 3	138	AlMalek El Nasser (16)	448
St. 4	148	AlMalek El Nasser North (C.C)	438
St. 5	161	El Dewayshat North (H - H)	422
St. 6	178	Kagnarty (6)	394
St. 7	187	Morshed (3)	378.5
St. 8	197	Second Cataract (26)	357
St. 9	200	Halfa Doghaim (24)	347
St. 10	201	Dabarosa (22)	337.5

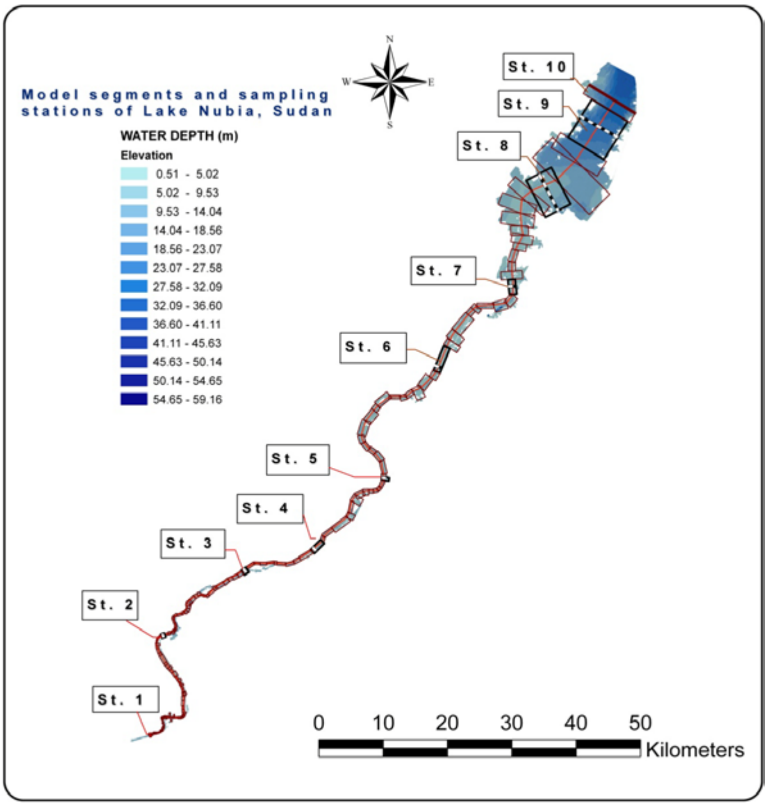

Figure 4.1: Lake Nubia model bathymetry January 2006, plan view.

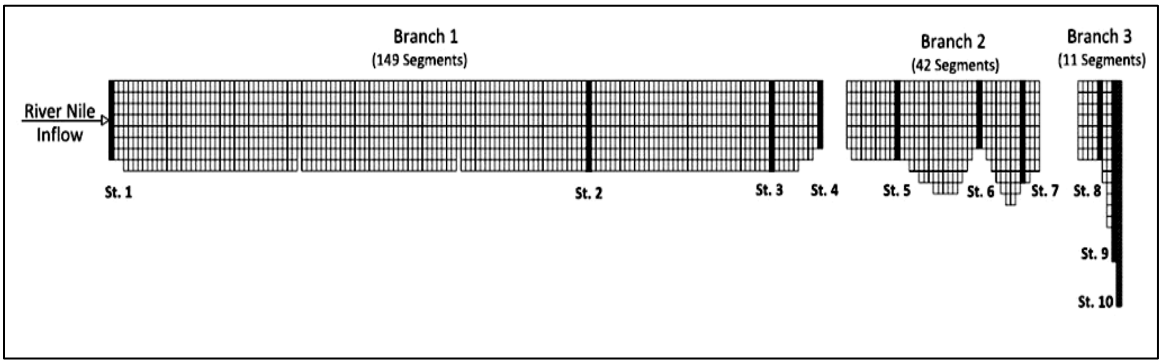

Figure 4.2: Lake Nubia model bathymetry January 2006, longitudinal section view.

4.2 Initial conditions

Initial conditions include number of water bodies (1), number of branches (3) and segments (202), bottom elevation of the reservoir (132 m AMSL), starting and ending times of the simulation, starting water surface elevation for each computational segment, initial temperature and water quality constituent concentrations.

The model simulation was set to begin on January 7, 2006 and finish on January 19, 2006 for calibration process, while for verification process, the model simulation was set to begin on February 2, 2007 and finish on February 14, 2007. These periods were selected because their data were the most complete available to develop the required files of the CE-QUAL-W2 model for Lake Nubia. These dates were converted to Julian Days, which is the required format for the model input files.

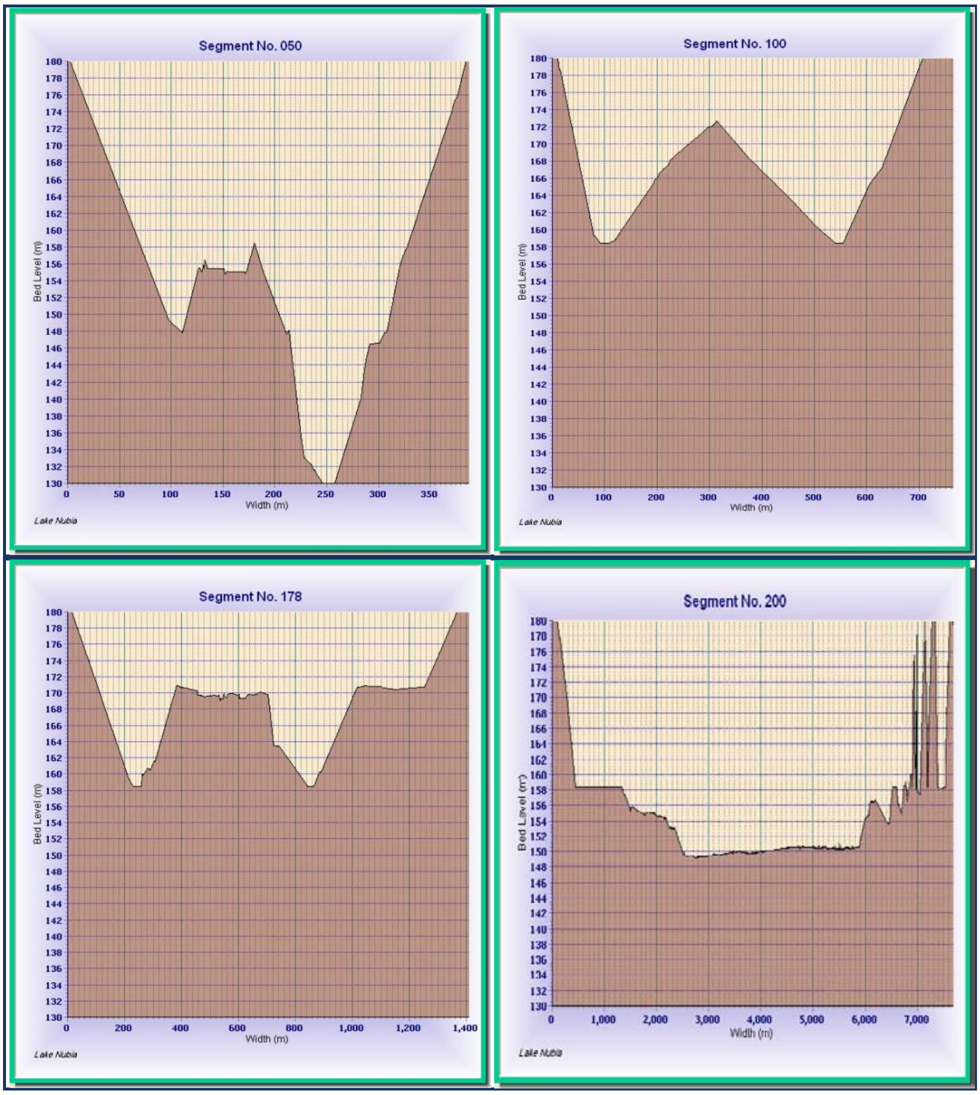

Figure 4.3: Lake Nubia model bathymetry January 2006, cross section view of some selected segments.

4.3 Meteorological data

The meteorological data were obtained for the nearby local meteorological station, Wadi Halfa station, from the internet (website of (Weather Underground)). The recorded data were given for every six hours. The meteorological data cover air temperature, dew point temperature, wind speed/direction and cloud cover. The station location with respect to the reservoir can be seen in Figure 4.4.

4.4 Boundary conditions

Lake Nubia CE-QUAL-W2 requires the following boundary conditions: upstream inflows (flow, water temperature and constituent concentrations) and downstream outflows (flow) or external downstream head in case of part of the reservoir is simulated.

The reservoir bottom is assumed to be an immobile and impermeable boundary, for which ground-water discharge to the reservoir or recharge from the reservoir to ground water is neglected. The reservoir shoreline is defined as a boundary across which there is no flow. The exact position of the shoreline changes during model simulation due to the change of water-surface elevation.

Lake Nubia has only one inflow source, River Nile, at its upstream end. The inflow data were specified as daily average values from one gauging station at the upstream end of the reservoir.

As mentioned before (section 4.1), ten control stations, St. 1 - 10 (Figure 4.1), were used to present Lake Nubia developed model results. The in-reservoir hydrodynamic and water quality conditions at the first station, St. 1 (Figure 4.1), were used as the upstream boundary conditions of the model.

Lake Nubia is the southern part of Aswan High Dam (AHD) reservoir. Therefore, the hydrodynamic and water quality characteristics of the reservoir at the last station, St. 10 (Figure 4.1), were used as the external downstream head of the model.

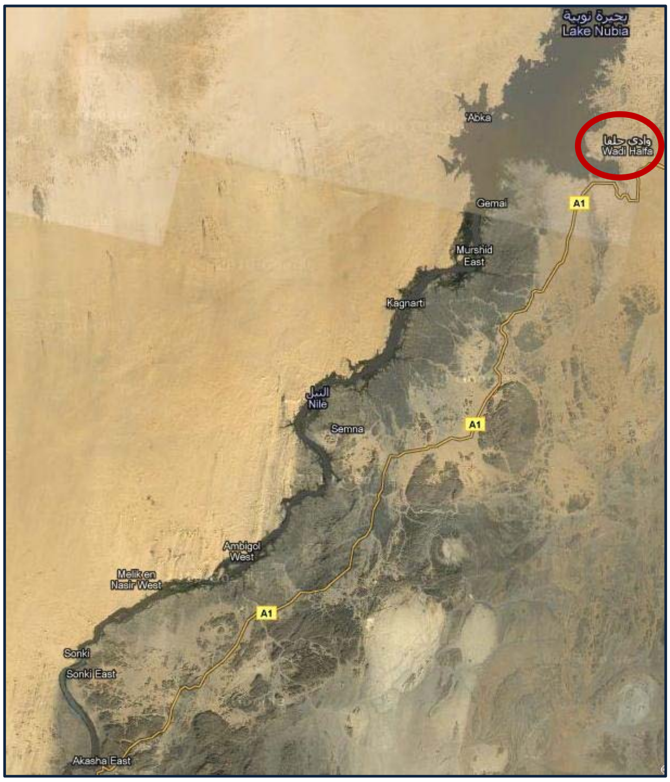

Figure 4.4: Location of Wadi Halfa meteorological station (Google maps).

4.5 Assumptions

The input data used for the model development are the best available and are supposed to be accurate representations of hydrological, meteorological, hydrodynamic and water quality parameters. The model accuracy may be also affected by the following additional assumptions:

4.5.1 Meteorological variability

As discussed in section 4.3, the meteorological data were obtained for the nearby local meteorological station, Wadi Halfa station. Wadi Halfa station is located at the northernmost end of Lake Nubia, as shown in Figure 4.4. The meteorological conditions at Wadi Halfa station may not represent conditions over the rest of the reservoir, especially near the major inflows and the southern end. There are no other weather stations close to the reservoir with observations as detailed as those at Wadi Halfa.

Model calibration includes adjusting for wind by using the "wind sheltering coefficient", a numerical value typically between 0 and 1 defined for individual segments (Cole and Wells, 2008). Wind speed values are multiplied by this coefficient to determine the effective wind speed for each model segment. The determination of wind sheltering coefficients for Lake Nubia was an iterative process and the values used in model simulations for individual segments were set to 1.0 as the study area is an open terrain.

4.5.2 Sediment transport

CE-QUAL-W2 code does not simulate processes of deposition or scour of sediments, although it simulates suspended soldis. From long-term observations of the AHD reservoir (Figure 4.5), the effect of sediment deposition becomes obvious, especially in the upper reaches of the Lake Nubia.

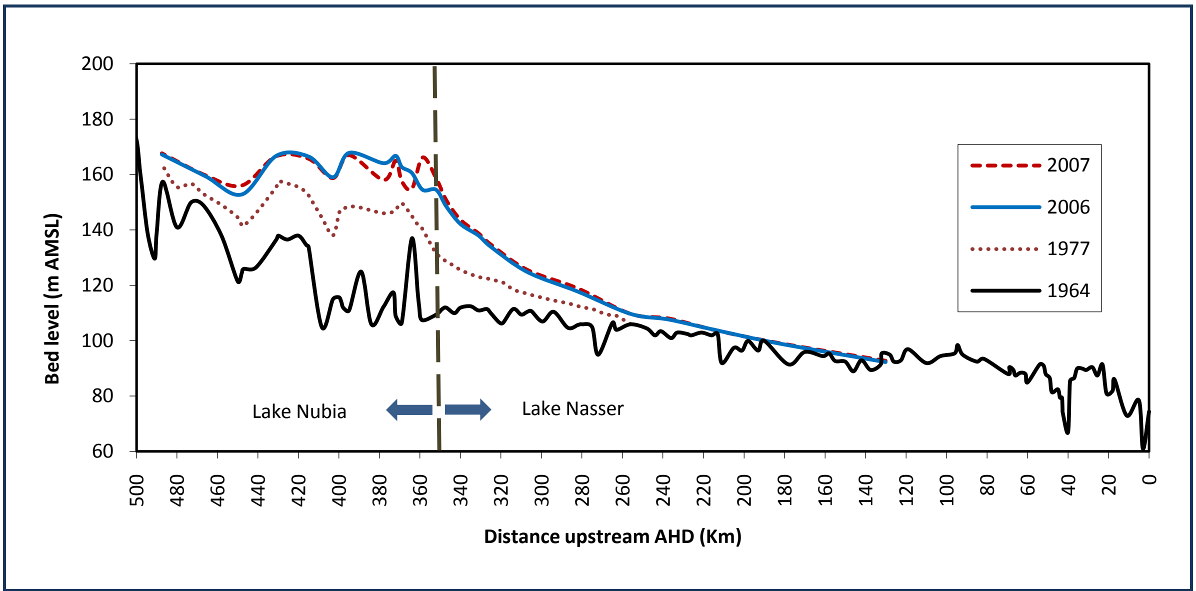

Figure 4.5: Long-term sediment accumulation of AHD reservoir.

4.5.3 Data gaps

The absence of data during certain periods may limit the ability to assess the accuracy of the calibration and verification of the model. Chlorophyll-a records have been measured at only two depths in the light and dark zones. The ammonium records were not available during the verification period. During both calibration and verification periods, BOD and IOC records were also not available.

4.6 Model calibration and verification

The proposed model was calibrated using coefficients determined for the calibration period (January 2006). For other model coefficients, which werenot available for this research work, default values recommended by Cole and Wells (2008) are used. Some model coefficients that affect the thermal calibration are displayed in Table 4.2.

As mentioned in section 3.1.5, two statistical indices were used to compare the simulated and in-reservoir measured observations; the absolute mean error (AME) and the root mean square error (RMS) (Cole and Tillman, 2001).

Table 4.2: Deferent model coefficients of CE-QUAL-W2 used in this research work.

Coefficient	Variable	Model value	Default
Horizontal eddy viscosity	AX	1.0 m^2/s	1.0 m^2/s
Horizontal eddy diffusivity	DX	1.0 m^2/s	1.0 m^2/s
Chezy bottom friction factor	CHEZY	70 $m^{½}/s$	70 $m^{½}/s$
Wind-sheltering coefficient	WSC	1.0	Parameter
Fraction of solar radiation absorbed at surface water	BETA	0.45	0.45
Sediment temperature	TSED	10 °C	-
Sediment heat exchange coefficient	CBHE	0.2 Wm^2/s	0.3 Wm^2/s
Interfacial friction	FI	0.015	0.010

The CE-QUAL-W2 model for Lake Nubia has been verified using collected data during February 2007. No model parameters were changed from those values established during the calibration process. The bathymetry has been modified due to sediment accumulation at some segments, as can be seen in Figure 4.5.

As Cole and Tillman (2001) pointed out that a point-to-point comparison of model predictions with observed data is the most rigorous mean of evaluating model output, vertical profiles of the simulated and the measured characteristics are presented in the following section (4.7).

4.7 Results and discussions

In this section, results of calibration and verification processes of the hydrodynamic and water quality characteristics of Lake Nubia are presented. The collected data used for calibration and verification processes are presented in the former chapter, while input files required for the current model are presented in previous sections of this chapter.

The investigated hydrodynamic characteristics are the surface water level and the water temperature, while water quality characteristics are the dissolved oxygen, chlorophyll-a, ortho-phosphate, nitrate-nitrite, ammonium, total dissolved solids, total suspended solids and pH.

The results are presented in this section for some stations. Appendix 2 includes the results at the other different stations. The presented results in this section were aggregately published by the author (Elshemy et al., 2010a; b; Elshemy and Meon, 2009).

4.7.1 Hydrodynamic results

4.7.1.1 Water surface levels

Figure 4.6 exhibits the longitudinal profile of water surface levels in Lake Nubia during the studied calibration period, in which a close match between the simulated and measured water surface levels can be noticed. AME is 0.088 m and the corresponding RMS is 0.104 m. For the verification process (Figure 4.7), AME is 0.136 m and RMS is 0.154 m. The AME and RMS values, of the calibration and verification processes, show good agreement of the simulated water surface levels with the observed water surface levels.

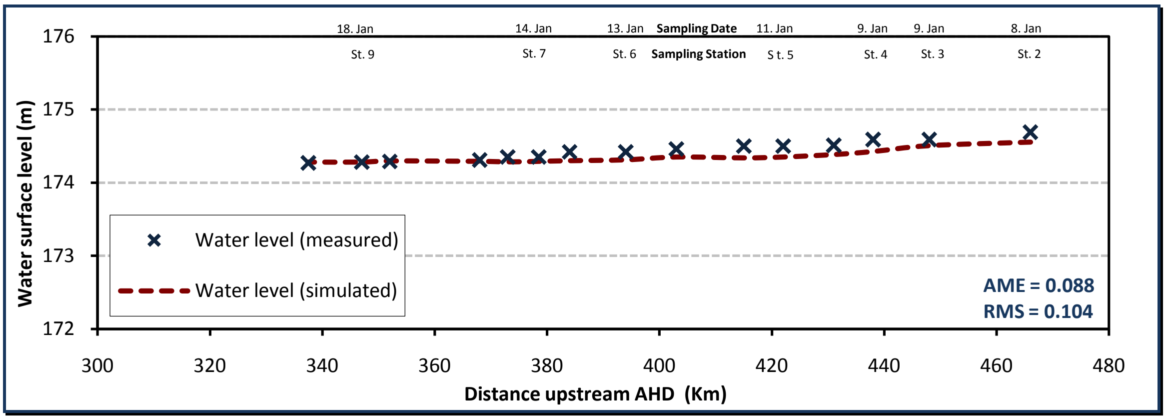

Figure 4.6: Longitudinal profile of water surface levels in Lake Nubia at different stations and dates, January 2006.

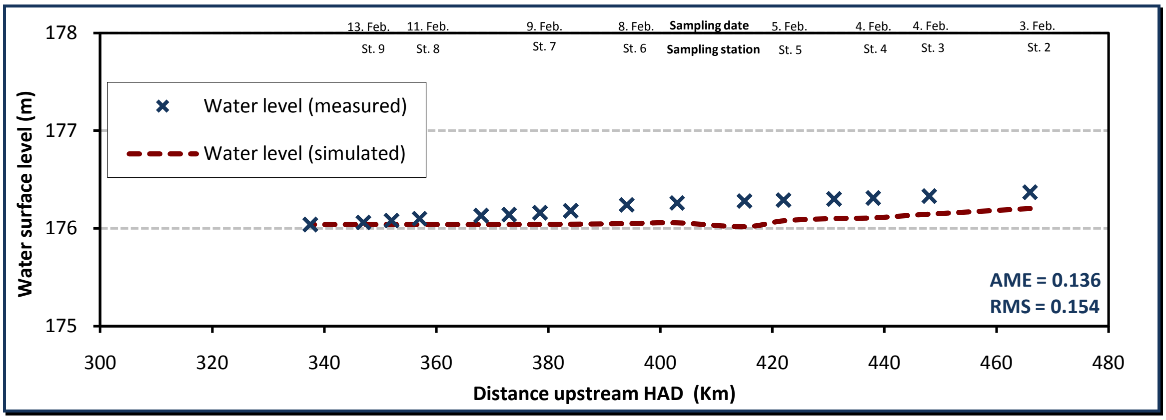

Figure 4.7: Longitudinal profile of water surface levels in Lake Nubia at different stations and dates, February 2007.

4.7.1.2 Thermal structure

The water temperature of Lake Nubia is affected by heat exchanges with the atmosphere and the water bottom, by inflows and outflows and heat generated by chemical and biological reactions. Water temperature affects the hydrodynamics of the reservoir. The inflows from River Nile, south of the reservoir at station No. 1 (St. 1), are generally warmer than reservoir water. The inflow water tends to spread out over the lake surface in the form of overflow. Overflows can contribute to water quality variations, as they add materials directly to the more productive surface zones of the lake.

The measured water temperature of Lake Nubia spatially ranged from 20.8 °C (at St. 1) to 15.5 °C (at St. 8) during the calibration period. During the validation period, the reservoir water temperature was uniform; the average value was about 16.7 °C. The corresponding simulated results are 20.8 °C (at St. 1) and 15.55 °C (at St. 8) during the calibration period, while the average simulated result during the validation period is 16.9 °C.

Vertical profiles of simulated and measured temperature at different stations for the calibration and verification processes are shown in Figures 4.8 and 4.9, respectively. The simulation results match closely the measured vertical temperature profiles. The average AME of the calibration period for the eight stations (St. 2–9) is 0.255 °C, while the corresponding average RMS is 0.273 °C. For the verification period, the average AME is 0.368 °C and the corresponding average RMS is 0.397 °C. These vertical profiles are identically with the typical thermal distribution for warm monomictic lakes in winter (Thomas et al., 1996).

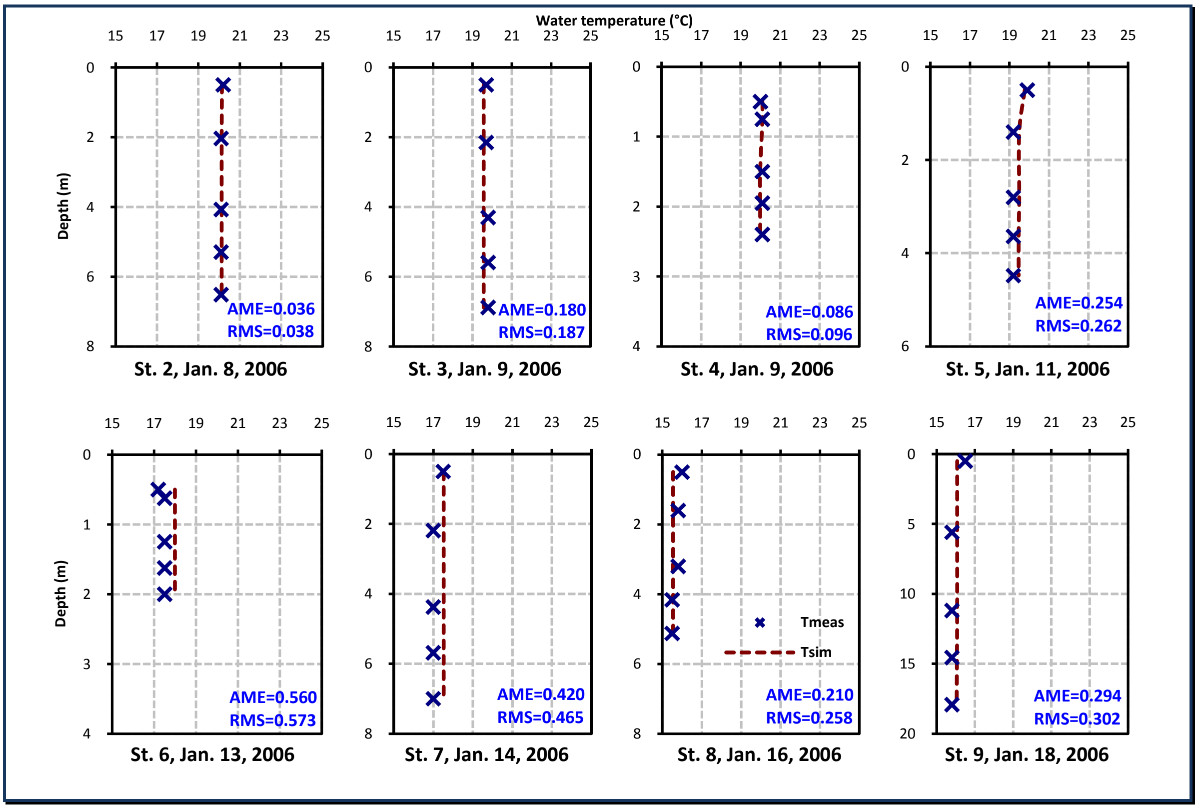

Figure 4.8: Model calibration: vertical profiles of water temperature in Lake Nubia at different stations and dates, January 2006.

4.7.2 Water quality results

The Lake Nubia model was calibrated and verified for dissolved oxygen, chlorophyll-a, ortho-phosphate, nitrate-nitrite, ammonium, total dissolved solids, total suspended solids and pH. The calibration and verification results are presented by comparison of simulated values to measured vertical profiles collected at 10 sites in the reservoir. Model calibration and verification were limited by the small number of measured records, which were available only during calibration and verification periods. Calibration and verification periods of Lake Nubia model are low flood periods occuring in January 2006 and February 2007, respectively.

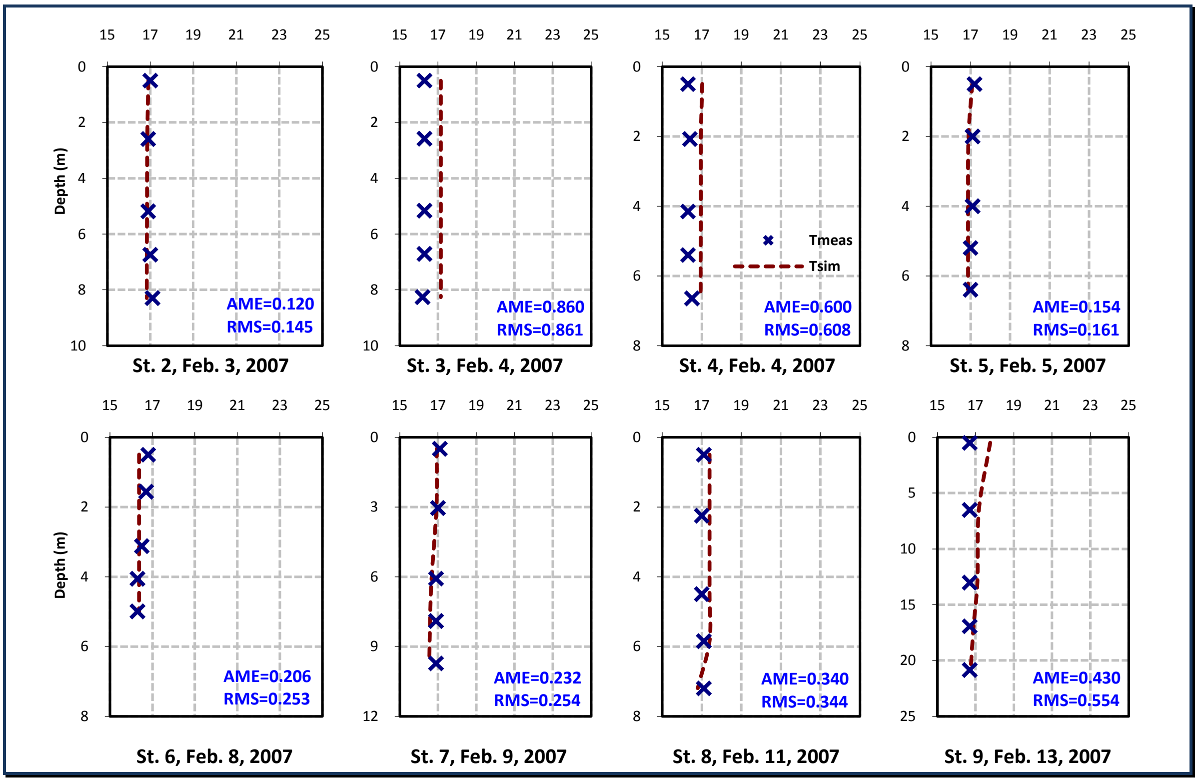

Figure 4.9: Model verification: vertical profiles of water temperature in Lake Nubia at different stations and dates, February 2007.

4.7.2.1 Calibration results

Average AME values and the corresponding RMS values for water quality parameters of Lake Nubia model during calibration period (January 2006) are presented in Table 4.3. The water quality model results and field measurements of the calibration process for stations No. 2, 4 and 8, as examples, are presented in vertical profiles, as shown in Figures 4.10, 4.11 and 4.12, respectively. Figure 4.13 presents an example of Lake Nubia model contour results for some selected parameters in Branch No. 3 on 16 January 2006

Table 4.3: Average water quality AME and RMS of Lake Nubia model calibration, January 2006.

Parameter	Unit	AME	RMS
Dissolved oxygen (DO)	mg/L	0.275	0.279
Chlorophyll-a (Chl-a)	mg/L	0.002	0.002
Ortho-phosphate (PO_4)	mg/L	0.010	0.013
Nitrate-nitrite (NO_3-NO_2)	mg/L	0.095	0.117
Ammonium (NH_4)	mg/L	0.029	0.035
Total dissolved solids (TDS)	mg/L	1.096	1.096
Total suspended solids (TSS)	mg/L	13.626	13.932
Potential of hydrogen (pH)	unit	0.799	0.805

Dissolved-oxygen (DO) concentrations simulated by the Lake Nubia model are influenced by algal photosynthesis and respiration, nitrification, decay of organic matter, aeration from interaction with the atmosphere, inflow and outflow DO concentrations and water temperature (Cole and Wells, 2008). The measured DO concentrations of Lake Nubia spatially range from 8.5 mg/L (at St. 7) to 8.76 mg/L (at St. 2) during the calibration period, while the simulated DO concentrations, during the same period, range from 8.54 mg/L (at St. 5) to 9.2 mg/L (at St. 9). The model presents higher concentrations of DO, than the measured values, at the downstream part of the reservoir due to the corresponding water temperatures decrease.

As can be seen from Figures 4.8 – 4.12, the vertical profiles of water temperature and dissolved oxygen closely match the typical winter mixing where the thermal pattern of the reservoir is warm monomictic; the reservoir stratifies in summer and mixing occurs in winter (Williams, 2005).

In general, the simulated dissolved-oxygen concentrations for the calibration period fitt very well to the measured concentrations. Spatially, the simulated concentrations slightly overpredict than the measured values at the downstream part of the reservoir than at other locations in the reservoir, may be because of the simplified algal dynamics which is defined in the model. Overall, the AME ranges from 0.045 (St. 5) to 0.593 mg/L (St. 7) and the RMSE ranges from 0.049 (St. 5) to 0.593 mg/L (St. 7).

Algal growth in the Lake Nubia model is affected by temperature, light and the availability of nutrients. Decreases in algal population in the model are due to mortality, respiration, excretion and settling to the bottom sediments (Cole and Wells, 2008).

Generally, as can be noticed from Figures 4.10 – 4.12, the simulated chlorophyll-a (Chl-a) concentrations are similar to the measured concentrations. Overall, the AME ranges from 0.000 (most of the stations) to 0.007 mg/L (St. 5) and the RMSE ranges from 0.000 (most of the stations) to 0.007 mg/L (St. 5). The difference between the simulated and the measured concentrations at St. 5 is due to the data gap; only one measured value can be used for comparison with the model results.

Nutrient concentrations in Lake Nubia model are affected by a number of processes. Nitrate is contributed by River Nile inflows and nitrification of ammonia and is consumed by algal uptake during growth. Sources of ammonium (NH_4) include River Nile inflows, algal respiration and decay of organic matter. Sinks include nitrification (conversion to nitrate), algal uptake during growth and reservoir outflow. Sources of ortho-phosphorus (PO_4) include River Nile inflows, algal respiration and decay of organic matter in the sediment and water column. Sinks include algal uptake during growth, settling of particles containing or absorbing phosphorus and reservoir outflow (Cole and Wells, 2008).

At Lake Nubia, vertically measured records of nitrate-nitrite (NO_3-NO_2) and ammonium (NH_4) at some stations may be uncertain, which affects the calibration assessment (Figures 4.11 and 4.12); for small depth differences, there are big nitrate-nitrite and ammonium concentration differences. In general, simulated nitrate-nitrite and ammonium concentrations for the calibration period coincide well with the measured concentrations (Figures 4.10-412). Overall, for nitrate-nitrite, the AME ranges from 0.021 (St. 7) to 0.236 mg/L as nitrogen (St. 4) and the RMSE ranges from 0.043 (St. 7) to 0.309 mg/L (St. 4). For ammonium, the AME ranges from 0.012 (St. 9) to 0.053 mg/L as nitrogen (St. 6) and the RMSE ranges from 0.016 (St. 9) to 0.066 mg/L (St. 6).

Generally, simulated ortho-phosphorus concentrations for the calibration period compare very well to the measured concentrations (Figures 4.10-412). Overall, the AME ranges from 0.002 (St. 8) to 0.022 mg/L as phosphorus (St. 5) and the RMSE ranges from 0.002 (St. 8) to 0.023 mg/L as phosphorus (St. 5).

Total dissolved solids (TDS) can affect water density and ionic strength, thereby affecting water movements, pH and the distribution of carbonate species (Cole and Wells, 2008). The simulated TDS generally are similar to the measured TDS concentrations. the AME ranges from 0.097 (St. 3) to 3.414 mg/L (St. 8) and the RMSE ranges from 0.097 (St. 3) to 3.414 mg/L (St. 8).

Total suspended solids (TSS) correspond to non-filterable residues which affect light penetration and productivity, turbidity, recreational values, habitat quality, pH and cause reservoir to fill in faster. The measured TSS spatially ranges from 138.6 mg/L (St. 1) to 25.2 mg/L (St. 9) during the calibration period, this decrease of TSS concentration is due to the morphological change of the reservoir which affects the settling velocity of TSS. The simulated TSS concentrations generally fit well to measured TSS concentrations (Figures 4.10-4.12). Overall, the AME ranges from 1.73 (St. 9) to 29 mg/L (St. 6) and the RMSE ranges from 2 (St. 9) to 29 mg/L (St. 6).

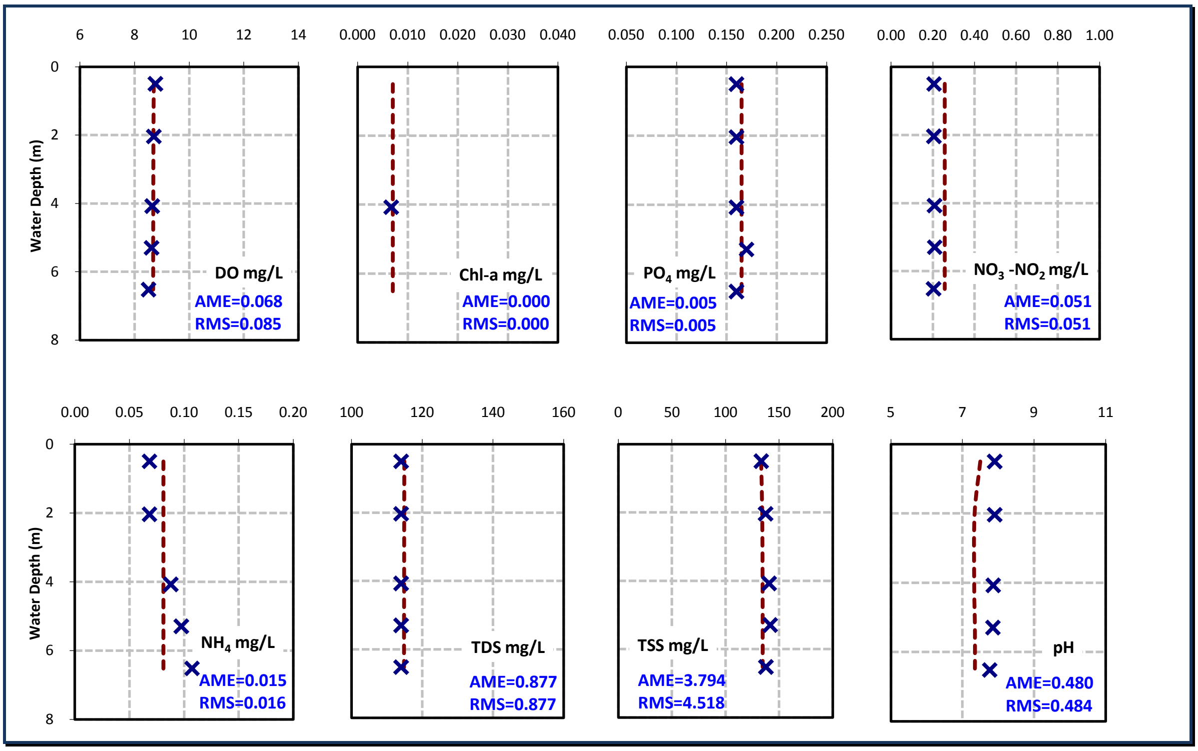

Figure 4.10: Model calibration: vertical profiles of measured (x) and simulated (---) key water quality parameters in Lake Nubia at St. 2 on 8. January 2006.

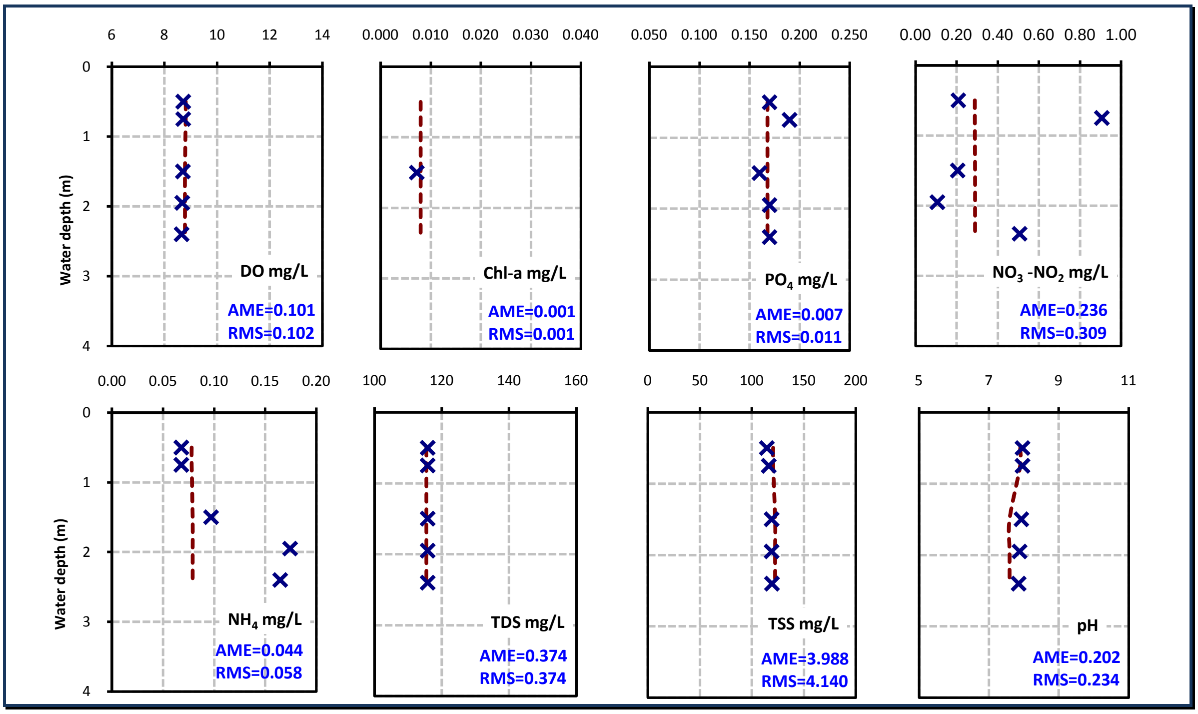

Figure 4.11: Model calibration: vertical profiles of measured (x) and simulated (---) key water quality parameters in Lake Nubia at St. 4 on 9. January 2006.

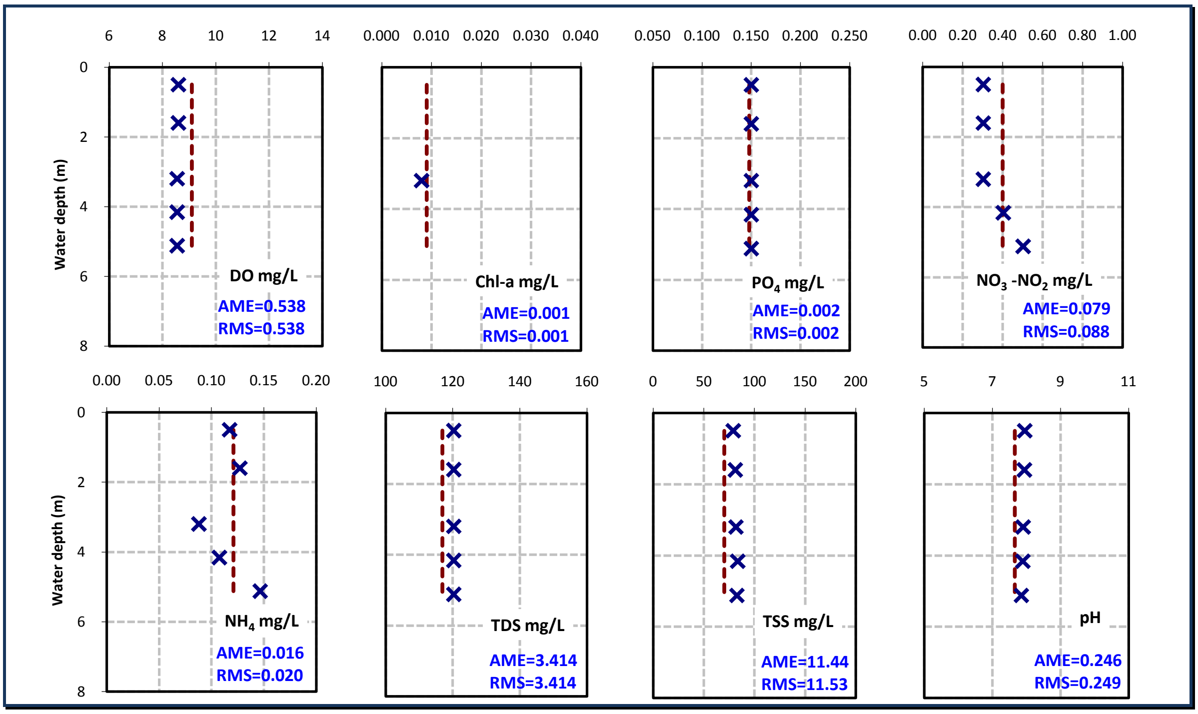

Figure 4.12: Model calibration: vertical profiles of measured (x) and simulated (---) key water quality parameters in Lake Nubia at St. 8 on 16. January 2006.

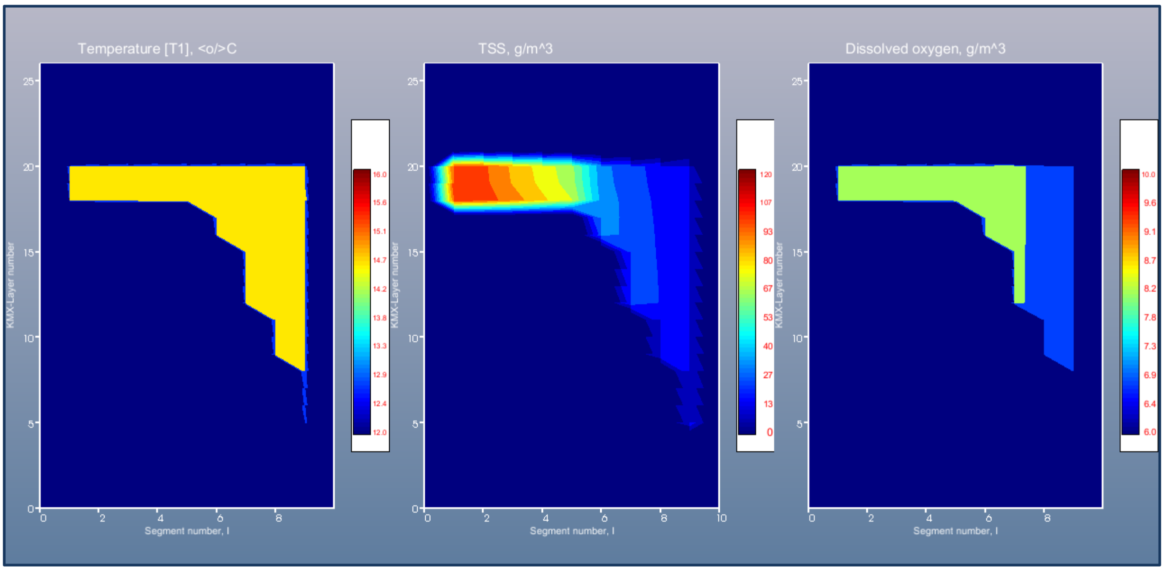

Figure 4.13: Model calibration: Lake Nubia model contour results for some selected parameters in Branch No. 3, on 16. January 2006.

4.7.2.2 Verification results

Average AME values and the corresponding RMS values for water quality parameters of Lake Nubia model during the verification period, February 2007, are presented in Table 4.4. The water quality model results and field measurements of the verification process for St. 4, St. 6 and St. 8, as examples, are presented in vertical profiles, as shown in Figures 4.14, 4.15 and 4.16, respectively.

Table 4.4: Average water quality AME and RMS of Lake Nubia model verification, February 2007.

Parameter	Unit	AME	RMS
Dissolved oxygen (DO)	mg/L	0.491	0.546
Chlorophyll-a (Chl-a)	mg/L	0.003	0.003
Ortho-phosphate (PO_4)	mg/L	0.009	0.009
Nitrate-nitrite (NO_3-NO_2)	mg/L	0.053	0.057
Ammonium (NH_4)	mg/L	N.A.*	N.A.*
Total dissolved solids (TDS)	mg/L	0.971	1.084
Total suspended solids (TSS)	mg/L	12.054	12.795
Potential of hydrogen (pH)	unit	0.643	0.675

*N.A.: There are no avilable field measurements.

During the verification period, measured ammonium concentrations, in general, are less than 0.010 mg/L as nitrogen but the measured records are not available.

Simulated DO and Chl-a concentrations compare better to measured concentrations during the calibration period than during the verification period. The other parameters, such as nitrate-nitrite, compare better during the verification period.

Figure 4.16 shows a decrease of chlorophyll-a and dissolved oxygen concentrations, at the bottom layer, and an increase of ortho-phosphate, nitrate-nitrite and total suspended solids as the phytoplankton growth is subjected to light limitation.

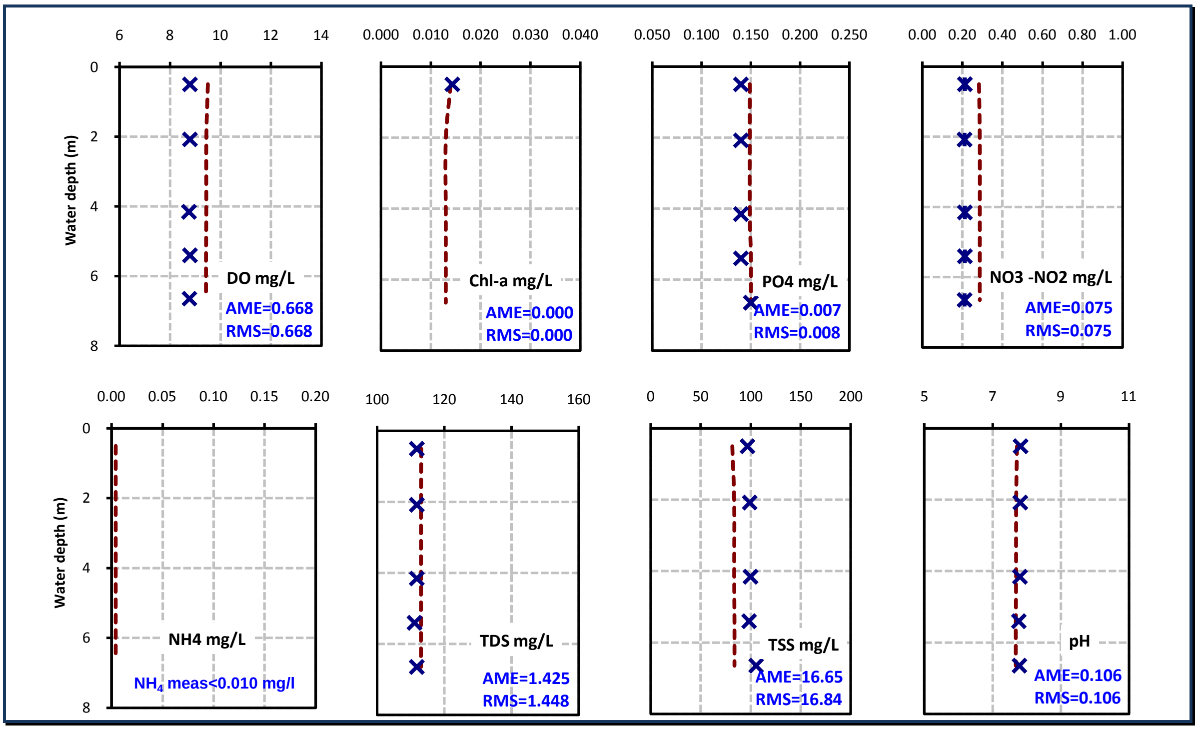

Figure 4.14: Model verification: vertical profiles of measured (x) and simulated (---) key water quality parameters in Lake Nubia at St. 4 on 11. February 2007.

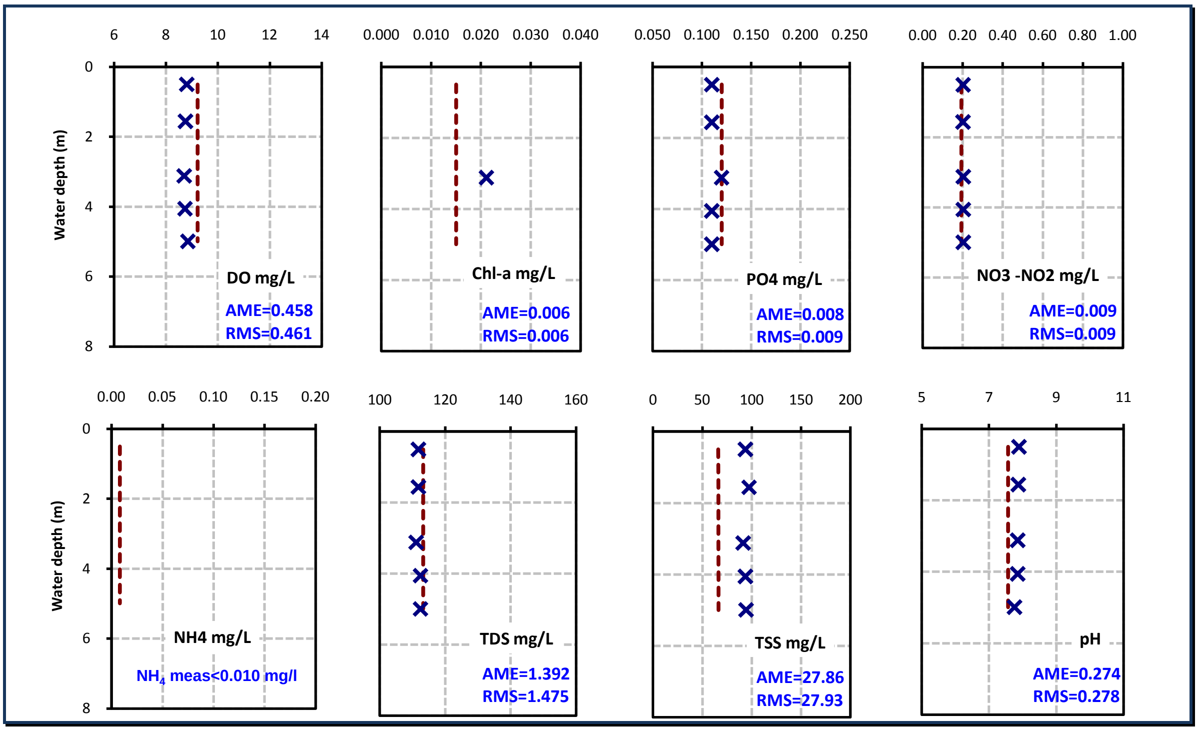

Figure 4.15: Model verification: vertical profiles of measured (x) and simulated (---) key water quality parameters in Lake Nubia at St. 6 on 11. February 2007.

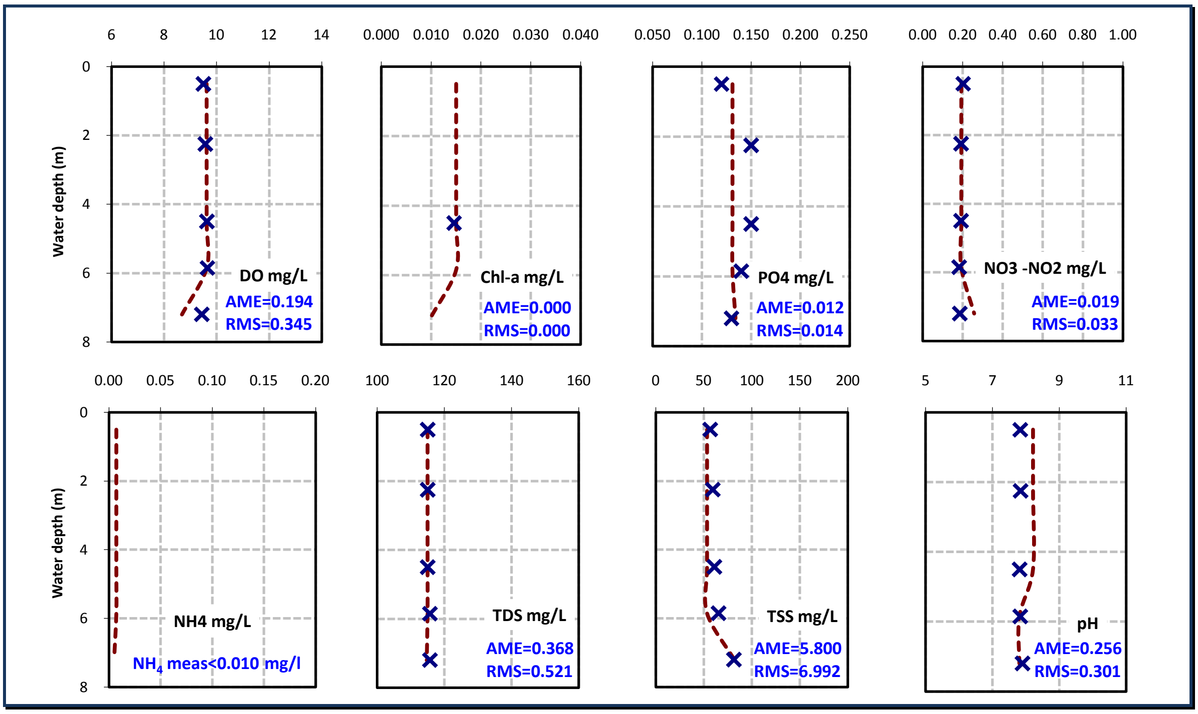

Figure 4.16: Model verification: vertical profiles of measured (x) and simulated (---) key water quality parameters in Lake Nubia at St. 8 on 11. February 2007.

An example of Lake Nubia model contour results for some selected parameters in Branch No. 3, on 12. February 2007, is presented in Figure 4.17.

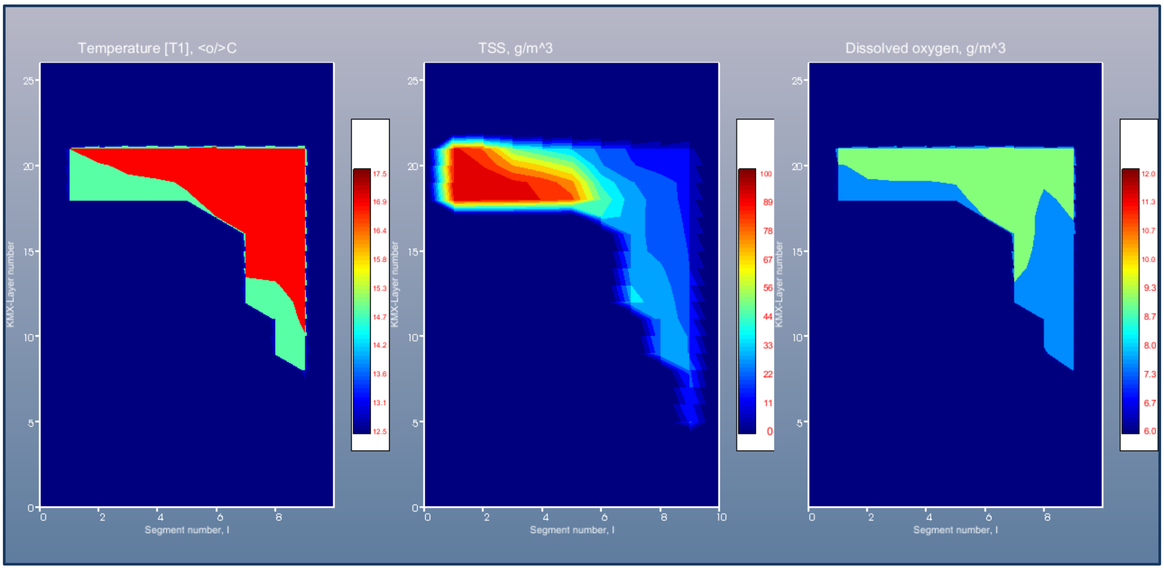

Figure 4.17: Model verification: Lake Nubia model contour results for some selected parameters in Branch No. 3, on 12. February 2007.

4.8 Conclusions

The present hydrodynamic and water quality characteristics of Lake Nubia are investigated by developing a laterally averaged, two-dimensional hydrodynamic and water quality modeling system (code), CE-QUAL-W2. The model was calibrated and verified with data from January 2006 and February 2007, respectively. The investigated hydrodynamic characteristics are the water surface level and the water temperature, while water quality characteristics are dissolved oxygen, chlorophyll-a, ortho-phosphate, nitrate-nitrite, ammonium, total dissolved solids, total suspended solids and pH.

Based on the presented results of the current research work, it is concluded that:

1. Generally, the hydrodynamic model simulation shows a good agreement with the observed water surface levels and the measured water temperature profiles at various locations and dates. The water quality model reproduces well the spatial and temporal concentration distributions of key water quality parameters. The model results closely mimic the measured vertical profiles of the water quality constituents.
2. The calibrated and verified Lake Nubia Model shows a good agreement of the simulated water surface levels with the observed water surface levels. For the calibration period AME is 0.088 m and the corresponding RMS is 0.104 m, while for the verification process AME is 0.136 m and RMS is 0.154 m.
3. For the thermal structure, the developed model results match closely the measured vertical temperature profiles. The average AME of the calibration period is 0.255 °C, while the corresponding average RMS is 0.273 °C. For the verification period, the average AME is 0.368 °C and the corresponding average RMS is 0.397 °C.
4. The vertical profiles (measured and simulated) of water temperature and dissolved oxygen closely match the typical thermal distribution for warm monomictic lakes in winter.
5. The simulated dissolved-oxygen concentrations for the calibration period coincide very well with the measured concentrations. Overall, the average AME is 0.275 mg/L and the corresponding RMS is 0.279 mg/L.
6. The simulated chlorophyll-a (Chl-a) concentrations are similar to the measured concentrations. The average AME and RMS are 0.002 mg/L.
7. The simulated nitrate-nitrite and ammonium concentrations for the calibration period compare well to the measured concentrations. For nitrate-nitrite, the average AME and RMS are 0.095 and 0.117 mg/L as N, respectively, while for ammonium, the average AME and RMS are 0.029 and 0.035 mg/L as N.
8. The simulated ortho-phosphorus concentrations for the calibration period compare very well to the measured concentrations. The average AME and RMS are 0.010 and 0.013 mg/L as P, respectively.

4.9 Recommendations for future work

The results of this research work have generated new options for future work which include:

1. Further development of the current modeling system which can be used as a management tool for the evaluation of different management strategies, such as climate and hydrologic forcing scenarios, of controlling the reservoir hydrodynamic and water quality characteristics.

2. Further collection of in-situ hydrodynamic and water quality data. The reservoir water quality monitoring program can use the model’s requirements and simulation results to determine the measurement frequency and the location of water quality parameters. Measured hydrodynamic and water quality records for the reservoir for – at least – a complete year should be provided to refine the developed model.
3. Locating several meteorological stations for providing the required meteorological data and refining the input meteorological conditions of the developed model.
4. Improvement of the algorithms of some of the physical, biological and chemical processes of the model by developing the modeling code to provide more accurate results.
5. Research on sediment transport and delta interactions.
6. Applying the concepts of model development to numerous other reservoirs along the River Nile.

5. Water quality assessment

5.1 Introduction

Water quality state of a water body depends on a large number of physical, chemical and biological indicators. An evaluation approach, such as water quality index that can be used to indicate the overall water quality condition, is necessary.

Basically there are four main approaches which are widely used to assess the water quality of a water body:

1. Water quality index approach.
2. Trophic status index approach.
3. Statistical analysis approaches of the water quality data such as correlation analysis (Boyacioglu and Boyacioglu, 2007; Cairns et al., 1997; Camargo et al., 2005; Debels et al., 2005; Dodds et al., 2006; Toufeek and Korium, 2009) and fuzzy clustering analysis (Kung et al., 1992; Liou and Lo, 2005; Lu and Lo, 2002; Lu et al., 1999; Ocampo-Duque et al., 2006).
4. Biological analysis approaches such as Genetic Algorithms method and other different biological indices (Ali et al., 1999; Bikbulatov and Stepanova, 2002; Bonanno and Lo Giudice, 2010; Fremling and Johnson, 1990; Huszar et al., 1998; Nuernberg, 1996; Peng, 2004; Rendón-Von Osten et al., 2006; Viaroli and Christian, 2003).

First, comprehensive studies are required to develop statistical and biological approaches. This task was not of the frame of this research work, but is recommended in future work. In this research work, water quality and trophic status indices will be used to assess the water quality state of Lake Nubia.

5.2 State-of-the-art

5.2.1 Water quality index

A water quality index (WQI) can be defined as any mathematical approach which aggregates data on two or more water quality variables to produce a single number (Ott, 1978). It consists of water quality variables such as dissolved oxygen, total phosphorus and fecal coliform, each of which has specific impacts to beneficial uses. The water quality index concept was first introduced more than 150 years ago, in 1848, in Germany where presence or absence of certain organisms in water was used as an indicator of the fitness or otherwise of a water source (Abbasi, 2002).

There are several water quality indices that have been developed to assess water bodies. In 1965, the first-ever modern WQI was developed and published by Horton (Horton, 1965). Horton's index uses ten variables, including commonly monitored ones such as dissolved oxygen, fecal coliform and temperature. It is computed as the weighted sum of sub-indices, which are calculated using a table of specific sub-index values corresponding to range of each variable.

Horton's index ranges from 0, representing poor water quality, to 100, representing perfect water quality (Ott, 1978).

Since 1965, many different water quality indices have been addressed in the literature (Brown et al., 1970; CCME, 2001; Deininger, 1980; Dinius, 1987; House, 1990; House, 1989; Kaurish and Younos, 2007; Liou et al., 2004; Said et al., 2004; Sarkar and Abbasi, 2006; Smith, 1990; Steinhart et al., 1982).

Two of the best known water quality indices, which have been frequently used, are the National Sanitation Foundation Water Quality Index (NSF WQI) and the Canadian Council of Ministers of the Environment Water Quality Index (CCME WQI). The two indices were chosen for this research work.

5.2.1.1 NSF WQI

The NSF WQI has been developed by the National Sanitation Foundation (NSF) in 1970 (Brown et al., 1970). A survey including 142 water quality scientists was conducted to conclude which water quality tests should be included in an index, out of about 35 tests. Nine water quality variables are used for the index: dissolved oxygen (DO), fecal coliform, pH, biochemical oxygen demand (BOD), temperature change, total phosphate, nitrate, turbidity and total solids. The index is computed as the weighted sum of sub-indices. Each parameter has a weight factor based on its test's importance in water quality (Table 5.1). A rating curve gives a sub-index quality value, which ranges from 0 to 100, corresponding to the field measurements (Figure 5.1 and Appendix 3). The NSF WQI can be calculated as follows:

$$NSF\ WQI = \frac{\sum_{i=1}^{n} W_i * Q_i}{\sum_{i=1}^{n} W_i} \tag{5.1}$$

where:

W_i Weight factor of the i^{th} parameter

Q_i Quality of the i^{th} parameter, can be obtained from the appropriate sub-index rating graph (Figure 5.1)

Table 5.1: NSF WQI weight factors

Parameter	Weight factor
Dissolved oxygen (%sat)	0.17
Fecal coliform (#/100 mL)	0.16
pH (standard units)	0.11
Biochemical oxygen demand (mg/L)	0.11
Temperature change (degrees C)	0.10
Total phosphate (mg/L)	0.10
Nitrates (mg/L)	0.10
Turbidity (NTU)	0.08
Total suspended solids (mg/L)	0.07

The WQI ranges have been defined as (Brown et al., 1970):

- 90-100 : Excellent
- 70-90 : Good
- 50-70 : Medium
- 25-50 : Bad
- 0-25 : Very Bad

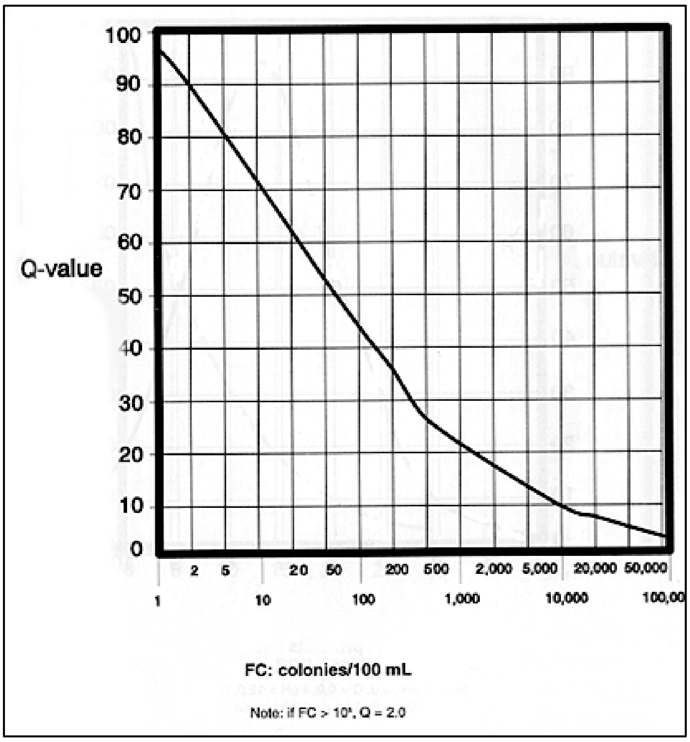

Figure 5.1: Fecal coliform rating graph (Wilkes University website).

The NSF WQI has been widely applied to a lot of water bodies (Abdel Rehim et al., 2002; Heikal et al., 2007; Sharma et al., 2006; Varnosfaderany et al., 2009; Wills and Irvine, 1996). NSF WQI, or variants of it, is used by a number of state agencies in the United States in their annual reports on the water quality of rivers and streams under their jurisdiction (Steinhart et al., 1982). NSF WQI was used to represent the water quality profile of Yamuna River, India (Sharma et al., 2008). In 2010, Bonanno and Lo Giudice used the NSF WQI and a floristic quality index. The former enabled them to describe the water quality according to a spatial–temporal gradient, whereas the latter focused on the ecological quality of riparian vegetation (Bonanno and Lo Giudice, 2010).

5.2.1.2 CCME WQI

In 1997, the Canadian Council of Ministers of the Environment (CCME) has developed a WQI to simplify the reporting of complex and technical water quality data. The WQI Technical Subcommittee adopted the conceptual model from the British Columbia index (CCME, 2001).

The application of the CCME WQI provides a measure of the deviation of water quality from water quality guidelines or objectives. Therefore, for each site and water use, different sets of parameters can be used depending upon the availability of data and regulatory standards.

The CCME WQI consists of three measures of variance from selected water quality objectives (scope, frequency, amplitude). These three measures of variance are combined to produce a value between 0 and 100 that represents the overall water quality. The CCME WQI values are then converted into rankings by using an index categorization schema that can be customized to reflect expert opinion by users. The detailed formulation of the WQI is described in the Canadian Water Quality Index 1.0 – Technical Report (CCME, 2001).

After the body of water, the period of time, the variables and the objectives have been defined, each of the three factors that make up the index must be calculated as follows:

F1 (scope) represents the percentage of variables whose objectives are not met in terms of "failed variables":

$$F_1 = \left(\frac{Number\ of\ failed\ variables}{Total\ number\ of\ variables}\right) * 100 \tag{5.2}$$

F2 (Frequency) represents the percentage of individual tests that do not meet objectives in terms of "failed tests":

$$F_2 = \left(\frac{Number\ of\ failed\ tests}{Total\ number\ of\ tests}\right) * 100 \tag{5.3}$$

F3 (Amplitude) represents the amount by which failed test values do not meet their objectives. F3 is calculated in three steps.

1. The number of times, by which an individual concentration is greater than (or less than, when the objective is a minimum) the objective, is termed an "excursion" and is expressed as follows.

When the test value must not exceed the objective:

$$excursion_i = \left(\frac{Failed\ test\ value_i}{Objective_j}\right) - 1 \tag{5.4}$$

For the cases in which the test value must not fall below the objective:

$$excursion_i = \left(\frac{Objective_j}{Failed\ test\ value_i}\right) - 1 \tag{5.5}$$

2. The collective amount by which individual tests are out of compliance. This variable, referred to as the normalized sum of excursions, or *nse*, is calculated as:

$$nse = \frac{\sum_{i=1}^{n} excursion_i}{Total\ number\ of\ tests} \tag{5.6}$$

3. F3 is then calculated by an asymptotic function that scales the normalized sum of the excursions from objectives *(nse)* to yield a range between 0 and 100.

$$F_3 = \left(\frac{nse}{0.01nse + 0.01}\right) \tag{5.7}$$

The CCME WQI is then calculated as:

$$CCME\ WQI = 100 - \left(\frac{\sqrt{{F_1}^2 + {F_2}^2 + {F_3}^2}}{1.732} \right) * 100 \tag{5.8}$$

The WQI values are then converted into rankings by using the index categorization schema as presented in Table (5.2). The rankings range from poor to excellent based on the WQI scores.

Table 5.2: CCMEWQI categorization scheme (CCME, 2001).

Rank	CCME WQI Score	Description
Excellent	95 – 100	Water quality is protected with a virtual absence of threat or impairment; conditions very close to natural or pristine levels. These index values can only be obtained if all measurements are within objectives virtually all of the time.
Good	80 – 94	Water quality is protected with only a minor degree of threat or impairment; conditions rarely depart from natural or desirable levels.
Fair	65 – 79	Water quality is usually protected but occasionally threatened or impaired; conditions sometimes depart from natural or desirable levels.
Marginal	45 – 64	Water quality is frequently threatened or impaired; conditions often depart from natural or desirable levels.
Poor	0 – 44	Water quality is almost always threatened or impaired; conditions usually depart from natural or desirable levels.

The CCME WQI has been applied to several data sets nationally (in Canada) and internationally (CCME, 2004; Donia and Farag, 2010; Ed Quilty, 2003; Lumb et al., 2006; Mercier et al., 2005; Radwan, 2005). In 2008, the CCME WQI was modified according to the Egyptian guidelines (for drinking surface water) and was applied at four stations of the River Nile, Egypt (Khan et al., 2008). De Rosemond et al. (2009) evaluated the ability of the CCME WQI to differentiate water quality from metal mines across Canada at exposure sites from reference sites using two different types of numeric water quality objectives. Two water quality indices (NSF WQI and CCME WQI) were trialed for the case of the Karun River system which is the most important river in Iran (Mojahedi and Attari, 2009). Boyacioglu (2009) has modified the CCME WQI to meet requirements of classification of surface waters according to quality based on European legislation. The modified CCME WQI has been applied to assess the overall water quality by integrating observed water quality determinants in the Kucuk Menderes Basin, Turkey. The CCME WQI was used to assess the water quality of the River Chenab, Pakistan, during the low-flow months of 2006–07 and 2007–08 (Bhatti and Latif, 2009).

5.2.2 Trophic status index

The Trophic state does not directly express the water quality but assigns the productivity of the water body due to the occurring biological and chemical activities in dependence on nutrient concentrations and other characteristics. Eutrophication process for Lakes and reservoirs has been discussed before in the second chapter (2.2.5.1). Lakes and reservoirs, according to their biological productivity and nutrient conditions, are commonly grouped into three different trophic states: oligotrophic, mesotrophic and eutrophic. The trophic state terms "oligotrophic" (Greek for 'little food') or "eutrophic" (Greek for 'well fed') were originally used to describe the soil fertility in northern Germany, and subsequently these terms were applied to lakes (Ryding et al., 1989).

Numerous attempts have been made to define the trophic states in terms of both nutrient and productivity water-quality parameters. Carlson (1977) has developed a numerical trophic state for lakes that incorporates most lakes in a scale of 0 to 100. The index number can be calculated from any of several parameters including Secchi disk transparency, chlorophyll and total phosphorus.

In 1982, a detailed classification system based on the German technical standards was developed and has been used successfully for lakes and reservoirs located primarily in temperate zones as well as in some non-temperate situations (Fachbereichsstandard, 1982; Ryding et al., 1989). This detailed classification system evaluates the water quality of a water body by using three main criterion classes: Hydrographic and territorial criterion, trophic criterion and salt content, special and hygienically relevant criterion (Ryding and Rast, 1989).

The Organization for Economic Cooperation and Development (OECD, 1982) provides specific criteria for temperate lakes in terms of the mean annual values of total phosphorus, chlorophyll-a and Secchi depth (Table 5.3). These criteria have limitations in its practical use; some water bodies can be classified in one or another trophic category depending on which parameter is used. Lake Baikal, Russia, for example, is ultra-oligotrophic in terms of physicochemical characteristics but is close to mesotrophic on the basis of primary production (Sigee, 2005). To avoid this, a more flexible open boundary system has been developed by OECD by using a statistical fit to more open ranges of the parameters. OECD criteria have been used for estimating the trophic status of several lakes and reservoirs (Dasí et al., 1998; Kasprzak et al., 2000; Koussouris et al., 1991; Lugliè et al., 2001; Schrenk-bergt et al., 2002).

The OECD concept has been modified by the German Regional Working Group on Water, Länderarbeitsgemeinschaft Wasser (LAWA, 2001). The LAWA Total Index (TI) is more flexible and can be calculated by different methods.

In the literature, many different trophic indices have been developed (Buetow, 2008; Liou and Lo, 2005; Lu and Lo, 2002; Lu et al., 1999; Novo et al., 1994; Nuernberg, 1996; Pavluk and Vaate, 2008; Peng et al., 2009; Salas and Martino, 1989 and 1991; Xu, 2008; Yun-feng, 2009; Zhang et al., 2008).

For reservoirs, based on their morphological characteristics, there is often a spatial gradient in sediment and nutrient concentration patterns along the body of a reservoir, especially in long, narrow, dendritic reservoirs. This gradient is accompanied by a spatial gradient in biological productivity and water quality in the reservoir (Ryding et al., 1989). The trophic status may range from nutrient rich (eutrophic) in the upper reaches of the water body to nutrient poor (oligotrophic) at locations closer to the dam wall (Thornton et al., 1996). Lind et al. (1993) suggested that according to the gradient in trophic status in reservoirs, a corresponding zonation exists in the relative suitability of various portions of reservoirs for different uses (e.g. water supply, fishing and recreational activities). Moreover, they stated that two different methods should be used for estimating the trophic status of reservoirs.

Table 5.3: Trophic classification of temperate freshwater lakes, based on a fixed boundary system (OECD, 1982; Sigee, 2005).

	Trophic category				
	Ultra-oligotrophic	**Oligotrophic**	**Mesotrophic**	**Eutrophic**	**Hyper-trophic**
Nutrient concentration (µg/L)					
Total phosphorus	<4	4-10	10-35	35-100	>100
Ortho-phosphate	<2		2-5	5-100	>100
DIN	<10		10-30	30-100	>100
Chlorophyll-a concentration (µg/L)					
Mean concentration in surface water	<1	1-2.5	2.5-8	8-25	>25
Max. concentration in surface water	<2.5	2.5-8	8-25	25-75	>75
Secchi depth (m)					
Mean annual value	>12	12-6	6-3	3-1.5	<1.5
Minimum annual value	>6	>3	3-1.5	1.5-0.7	<0.7

Two trophic status indices were chosen for this reserarch work; Carlson TSI and LAWA TI. In the forthcoming sections, Carlson TSI and LAWA TI are discussed.

5.2.2.1 Carlson TSI

The concept of trophic status is based on the fact that changes in nutrient levels (measured as total phosphorus) cause changes in algal biomass (measured as chlorophyll a) which in turn cause changes in lake clarity measured as Secchi disk transparency (Pavluk and Vaate, 2008). The Carlson TSI clearly represents this relationship. It offers the most suitable and acceptable method for trophic classifications of lakes (Xu, 2008). The Carlson TSI is independently calculated from Secchi depth (SD), chlorophyll a concentration (CHL) and total phosphorus concentration (TP).

The Carlson TSI can be calculated by using one of the following equations (Carlson, 1977; Carlson and Simpson, 1996):

$$TSI(SD) = 60 - 14.41\ ln[SD\ (m)] \tag{5.9}$$

$$TSI(CHL) = 9.81\ ln[CHL\ (\mu g l^{-1})] + 30.6 \tag{5.10}$$

$$TSI(TP) = 14.42\ ln[TP\ (\mu g l^{-1})] + 4.15 \tag{5.11}$$

According to the TSI value, lakes or reservoirs can be trophically classified as one of the following classes:

- TSI < 40 Oligotrophic
- 40 – 50 Mesotrophic
- 50 – 70 Eutrophic
- TSI > 70 Hypereutrophic

If the three TSI values are not similar to each other, Pavluk and Vaate (2008) return this to the possibilities that the algal growth may be light-limited or nitrogen-limited instead of phosphorus-limited or, among other reasons, that the Secchi disk transparency is affected by erosional silt particles rather than by algae. They recommend using the average of TSI values as an indicator of the water trophic status in general.

The Carlson TSI has been widely used to evaluate a lot of lakes and reservoirs (Arfi, 2003; Hassan, 2000; Koussouris et al., 1991; Mullins and Whisenant, 2004; Thiemann and Kaufmann, 2000). Galloway and Green (2004) used TSI(TP) and TSI(CHL) to assess the trophic status of lakes Maumelle and Winona, Arkansas, USA. The Florida Department of Environmental Protection has developed a new trophic status index, Florida Trophic State Index, which is based on the Carlson TSI but also includes total nitrogen as a third indicator (Paulic et al., 1996). For Florida water quality assessment reports, the Secchi depth has been excluded as an indicator because of the associated problems which have been caused in dark-water lakes and estuaries, for which dark waters rather than algae diminish transparency. In 2010, Santhanam and Amal Raj (2010) have used the Carlson TSI to determine the trophic status of Pulicat lagoon, India, over the years 2005 and 2006. All the values obtained for the TSI (SD) were higher when compared to the TSI (Chl-a), which may indicate that something other than algae, perhaps colour or non-algal seston or the dominance by pico-plankton, is contributing to the light attenuation. The investigation points out that there is a need to develop more modern and more accurate indices to represent water quality in a lagoon.

5.2.2.2 LAWA TI

The German Regional Working Group on Water LAWA, Länderarbeitsgemeinschaft Wasser, has modified the OECD concept to develop a new trophic index which meets the current German conditions (LAWA, 2001). A data base of 117 reservoirs in Germany was used. The index depends on three parameters; chlorophyll-a (Chl-a), Secchi depth (SD) and total phosphorus (P_S for summer and P_F for spring). The LAWA TI depends on the parameters sub-indices, which should be calculated as follows (LAWA, 2001; TLUG, 2004):

$$Index\ Chl - a = 0.560 + 0.856\ ln[Chl - a\ (\mu g l^{-1})] \tag{5.12}$$

For other parameters sub-indices, the equations vary according to reservoir morphology. Deep reservoirs, with maximum water depth between 14 and 77.5 m, parameters sub-indices are calculated as follows:

$$Index\ SD = 3.739 - 1.27\ ln[SD\ (m)] \tag{5.13}$$

$$Index\ P_F = -0.155 + 0.813\, ln[P_F\ (\mu g l^{-1})] \tag{5.14}$$

$$Index\ P_S = -0.939 + 1.066\, ln[P_S\ (\mu g l^{-1})] \tag{5.15}$$

Consequently, the LAWA Total Index (LAWA TI) can be calculated as follows:

$$LAWA\ TI = 0.939 + 0.285(Index\ Chl - a) - 0.301(Index\ SD) + 0.136(Index\ P_F\ ) + 0.249(Index\ P_S\ ) \tag{5.16}$$

For small reservoirs, maximum water depth < 13.5 m, parameters sub-indices are calculated as follows:

$$Index\ SD = 3.607 - 0.984\, ln[SD\ (m)] \tag{5.17}$$

$$Index\ P_F = 0.014 + 0.803\, ln[P_F\ (\mu g l^{-1})] \tag{5.18}$$

$$Index\ P_S = 0.548 + 0.722\, ln[P_S\ (\mu g l^{-1})] \tag{5.19}$$

Then the LAWA Total Index (LAWA TI) can be calculated as follows:

$$LAWA\ TI = 1.279 + 0.285(Index\ Chl - a) - 0.262(Index\ SD) + 0.134(Index\ P_F\ ) + 0.168(Index\ P_S\ ) \tag{5.20}$$

If the parameters sub-indices values are not similar to each other, another flexible method of the LAWA TI calculation can be used, in which the irregular parameter sub-index can be excluded. This method of calculation depends on parameter weight factors (Wf) according to Table 5.4.

Table 5.4: LAWA TI parameter weight factors

Parameter	Weight Factor (Wf)
Chlorophyll – a (Chl)	10
Secchi depth (SD)	8
Total phosphorus – spring (P_F)	5
Total phosphorus – summer (P_S)	7

Then LAWA TI can be calculated as follows:

$$LAWA\ TI = \frac{(Index\ Chl - a)\ Wf_{Chl} + (Index\ SD) Wf_{SD} + (Index\ P_F\ ) Wf_{PF} + (Index\ P_S\ ) Wf_{PS}}{\sum Wf} \tag{5.21}$$

According to LAWA TI value, lakes or reservoirs can be trophically classified as one of the following classes, Table 5.5.

LAWA TI has been widely used in Germany. It has been used to assess twenty three reservoirs in Saxony (Ackermann et al., 2005). The Rappbode dam reservoir in Saxony-Anhalt has been evaluated using LAWA TI. The reservoir trophic status was mesotrophic; LAWA TI was less than 2.0 (Schöpfer et al., 2007). The Ruhrverband (2008) used the LAWA TI to assess eight reservoirs in different sites in Germany. The Hessian Agency for Environment and Geology (HLUG, 2008) has applied LAWA TI to ninety water bodies (lakes and reservoirs) in Hessen for the water quality measurement in 2007. Scharf (2008) used the LAWA TI to evaluate the Wupper Reservoir which is situated in central western Germany.

Table 5.5: LAWA TI trophic status category

Deep reservoirs		Small reservoirs	
Trophic status	**LAWA TI**	**Trophic status**	**LAWA TI**
Oligotrophic	0.5 – 1.5	Eutrophic 1	2.6 – 3.0
Mesotrophic	1.6 – 2.5	Eutrophic 2	3.1 – 3.5
Eutrophic 1	2.6 – 3.0	Polytrophic 1	3.6 – 4.0
Eutrophic 2	3.1 – 3.5	Polytrophic 2	4.1 – 4.5
Polytrophic 1	3.6 – 4.0	Hypertrophic	4.6 – 5.0

5.3 Case study: Lake Nubia

5.3.1 Historical studies

The addressed studies regarding water quality evaluation of Aswan High Dam (AHD) reservoir in the literature are limited. Most of these studies were based on statistical approaches (Abdelbary et al., 1997a; Abdelbary et al., 1997b; Heikal et al., 1998; Heikal and Abdelbary, 1999).

In 1995, Awadallah and Soltan used a statistical approach to follow up the distribution of physical and chemical components between surface and bottom water of the AHD reservoir and their effects on the water quality and on the life of biota (Awadallah and Soltan, 1995). The samples were collected between Daal Cataract in Sudan and the AHD in the most southern part of Egypt, during the period from 23 November to 18 December 1992, at five different water depths. The statistical analysis of the database exhibited positive and significant correlation coefficient values. The results show that there was stratification in the water column of the reservoir.

The NSF WQI has been used by Abdel-Rehim et al. (2002) during the stratification period, extending from May to July, and turnover period, extending from September to December. The results revealed that the quality of water in Lake Nasser is improved during turnover and mixing periods. Average NSF WQI results were ranged between 62.00, medium, in July to 79.64, good, in November. The measured data were collected from one site in Lake Nasser, Abu Simbel (281 km upstream AHD), for one year (September 2000 to August 2001), and at different depths.

Abou El Kheir et al. (2003) studied the seasonal variations of physical-chemical characteristics and phytoplankton growth at seven stations along Lake Nasser during the period 2001- 2002. They stated that phosphorus is the limiting nutrient around the year. The Carlson TSI was used. The results show that Lake Nasser is a mesotrophic lake, its average TSI values range from 40 - 60. The TSI based on CHL was considerably lower than the TSI based on SD, especially in the south part of the lake. These results indicate that particles other than phytoplankton may be contributing to the light attenuation, where in this part of the lake, the water is turbid due to suspended sediment because of the flood or due to re-suspension from the bottom by turnover process.

A statistical mathematical model, based on regression relations of water quality parameters, was built up to estimate daily water quality parameters (Hussein and Shafi, 2005). A fixed laboratory at the entrance of the reservoir was assumed using the available data (1977 – 2001). A physical, chemical and biological analysis of the water quality was performed at 25 sections distributed along the lake using the World Health Organization (WHO) standards. According to WHO standards, the average water quality results before the flood are good, while during the flood are satisfactory except for the turbidity.

Gharib and Abdel- Halim (2006) used statistical regression models to study the species composition, abundance and phytoplankton biomass, supported by some physico-chemical environmental parameters, in Lake Nasser during the highest flood season in autumn 1999. The results showed that the most relevant physico-chemical factors affecting the growth of phytoplankton abundance were pH values, dissolved phosphate and reactive silicate.

Heikal et al. (2007) investigated the temporal and spatial variation of water quality status along the River Nile and the agricultural drains, which are the main sources of pollution along the Nile, during high and low flow periods from 2000 to 2005. Statistical analysis and NSF WQI were used. The results showed that the overall water quality status of the River Nile and the agricultural drains during the low flow period is generally better than during the high flow period. They concluded that the temporal and spatial variations in the water quality parameters along the River Nile are mainly affected by the quality of water discharges from agricultural drains as well as the magnitude of the River Nile flow.

In 2009, Toufeek and Korium (2009) studied the variations of physicochemical parameters in the main channel of Lake Nasser during the year 2005. Results indicated wide variations in the concentrations of different physicochemical parameters between surface and bottom layers during summer season especially in the northern part of the Lake. During winter, the variation in concentrations of these parameters between surface and bottom layers was modest. Correlations coefficient matrices between each two pairs of parameters were estimated to throw light on relationships between different physicochemical parameters. It was concluded that the various parameters under investigation in different seasons and regions in Lake Nasser lie within the permissible range, and the reservoir water quality status for drinking, irrigation and fish culture purposes is good.

Heikal (2010) investigated the quality of water in the main side branches (Khors) of Lake Nasser and its main channel during periods of low and high water levels for the years 2001 and 2005, respectively. He has used the NSF WQI and Carlson TSI to assess the water quality and trophic states, respectively, of Lake Nasser and its main Khors. The results showed that the reservoir water level drop led to a decline in the water quality state of Lake Nasser and its Khors from the order of good to medium. Also the trophic state index (TSI) values revealed that the productivity of the Lake changed from mesotrophic during the high water level season to eutrophic during the low water level season.

5.3.2 Research deficits and demands for this study

NSF water quality index (WQI) and Carlson trophic status index (TSI) are the most frequently used indices for Egyptian water resources, see the previous section. Here in this research work, a comparative study has been done by applying other different two indices (additional to NSF WQI and Carlson TSI). These indices are: CCME WQI and LAWA TI. Such the comparative study is essential to evaluate the water quality and trophic status for reservoirs in particular (Lind et al. (1993). CCME WQI was developed according to the Egyptian water quality standards for surface waterways. Carlson TSI was calculated for each of two water quality parameters, total phosphorus and chlorophyll-a, while the average Carlson TSI was calculated using both parameters, as recommended for reservoirs by Lind et al. (1993). Moreover, the developed hydrodynamic and water quality model of the investigated case study was validated using the results of these four applied indices.

5.3.3 Application of water quality and trophic status indices to Lake Nubia

Four indices have been developed to evaluate the water quality and trophic status of the southern part of AHD reservoir (Lake Nubia) during low flood period, January 2006. Two of these indices are the water quality indices NSF WQI and CCME WQI (described in the previous section). The other two indices are the trophic status indices Carlson TSI and LAWA TI. The measured data were collected during low flood period, January 2006, from ten control stations along Lake Nubia at different water depths, as explained in detail in the previous chapters. A comparative study has been done to the measured data by using the results of a developed hydrodynamic and water quality model, as discussed in detail in the previous chapter, during the calibration process.

5.3.3.1 NSF WQI

According to the available water quality parameters measurements of Lake Nubia, eight parameters were used to apply NSF WQI to Lake Nubia. The used parameters are: dissolved oxygen (% sat), fecal coliform (colonies/100 ml), pH (standard unit), temperature change (degrees C), total phosphate (mg/L), nitrate (mg/L), turbidity (NTU) and total solids (mg/L).

For temperature change, which is calculated as the difference between the water temperature value of the intended control station and water temperature value of the reference station upstream the water body, control station No. 1 (St. 1) has been used as a reference station. An expression for the dissolved oxygen saturation concentration (DO_{sat}) at sea level for freshwater as a function of water temperature which is given in APHA (Clescerl et al., 1999), Eq. 5.22, was used to calculate DO (% sat) at different control stations, Eq. 5.23. The used expression was as follows:

$$ln\,DO_{sat} = -139.34411 + \frac{1.575701 * 10^{5}}{T_w} - \frac{6.642308 * 10^{7}}{T_w^2} + \frac{1.243800 * 10^{10}}{T_w^3} - \frac{8.621949 * 10^{11}}{T_w^4} \quad (5.22)$$

And then:

$$DO\ (\%\ sat) = \left(^{DO_i}/_{DO_{sat}}\right)\% \tag{5.23}$$

where:

DO_{sat}	Freshwater DO saturation concentration in mg/L at sea level
T_w	Water temperature in (°K) ([°K]=[°C]+ 273.150)
DO_i	Measured or simulated DO concentration at the control station St. i

Because CE-QUAL-W2 model cannot simulate turbidity, for simulated water quality parameters, a linear regression analysis for the measured values of turbidity and total solids, January 2006, has been done (Figure 5.2). The resulted formula was used to calculate the turbidity based on simulated total solids (Eq. 5.24).

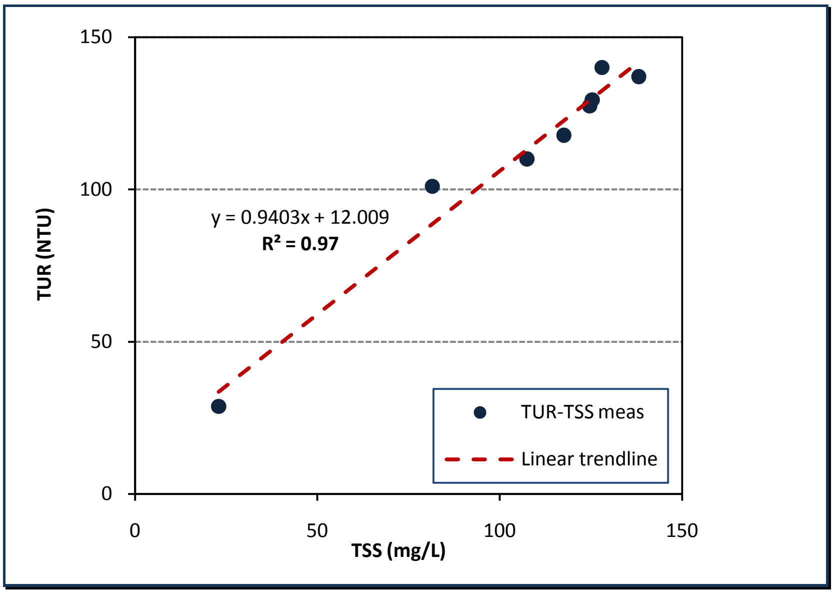

Figure 5.2: Measured turbidity – total solids linear regression analysis, January 2006.

$$TUR\ (NTU) = 0.9403 * TSS + 12.009\ \ ,R^2 = 0.97 \tag{5.24}$$

The measured and simulated water quality parameters used to obtain NSF WQI as well as the Q-values of the parameters and the resulted NSF WQI values are listed in Appendix 3 (a sample calculation).

Figure 5.3 shows the longitudinal profiles of the developed NSF WQI along Lake Nubia at different stations for measured and simulated water quality parameters. The absolute mean error (AME), for NSF WQI based on measured and simulated water quality parameters, is 2.1. The results show that the water quality status of Lake Nubia, according to NSF WQI, is good. The water quality status of the reservoir varies spatially; NSF WQI increases from 80 at St. 2 to 86 at St. 9. This increase in the water quality returns to the decrease of water turbidity and fecal coliform concentration.

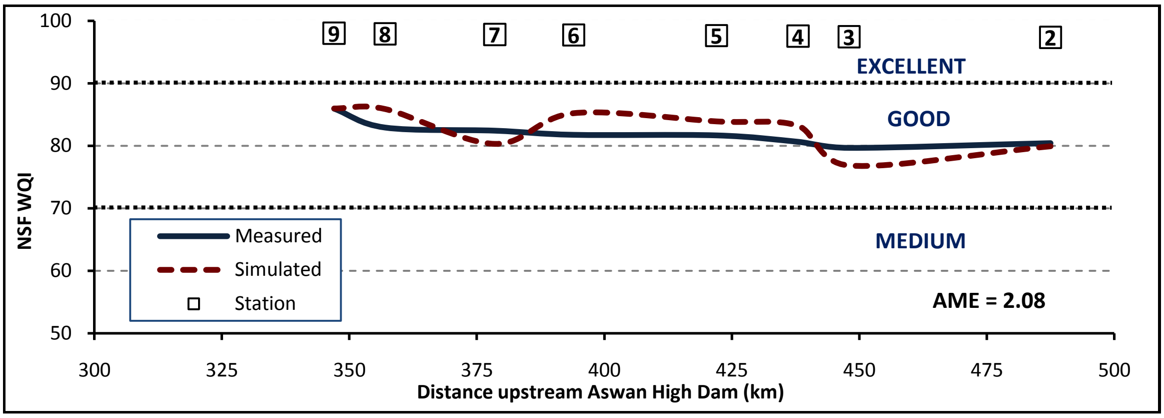

Figure 5.3: Lake Nubia NSF WQI longitudinal profiles at different stations for measured and simulated water quality parameters, January 2006.

The average NSF WQI of Lake Nubia for measured and simulated water quality parameters are shown in Figure 5.4. The developed hydrodynamic and water quality model shows good agreement with the measured parameters; the average AME is 0.4. The results indicate that the overall water quality status of Lake Nubia is good.

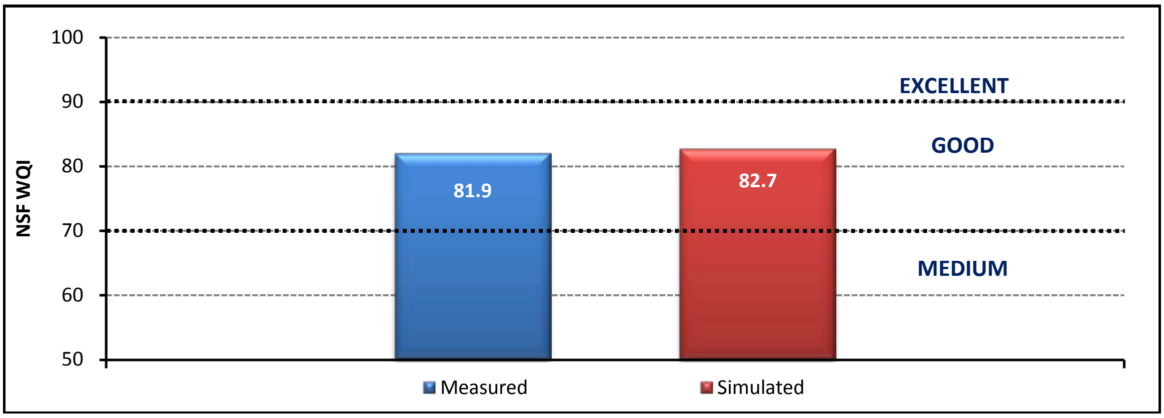

Figure 5.4: Lake Nubia average NSF WQI for measured and simulated water quality parameters, January 2006.

5.3.3.2 CCME WQI

The CCME WQI was developed to estimate the overall water quality status of Lake Nubia. Seven water quality parameters, measured and simulated (January, 2006), have been used. These parameters are: Dissolved oxygen (mg/L), total dissolved solids (mg/L), nitrate-nitrite (mg/L), ammonium (mg/L), total phosphorus (mg/L), fecal coliform (colonies/100 ml) and pH (standard unit).

The CCME WQI has been developed according to the Egyptian water quality standards for surface waterways (objectives), Law 48/1982 – Article No. 60 (EEAA, 1991), see Table 5.6.

Table 5.6: Egyptian water quality standards for surface waterways (objectives)

Parameter	DO (mg/L)	TDS (mg/L)	NO_3 (mg/L)	NH_4 (mg/ L)	TP (mg/L)	FC (N/100 mL)	pH (unit)
Objective	>5	<500	<45	<0.5	<1	<2000	7<...<8.5

For fecal coliform, as there is no Egyptian standard for it, the used objective was previously used by Heikal et al. (2007).

Figure 5.5 shows the overall CCME WQI of Lake Nubia for measured and simulated water quality parameters, January 2006. The results show that the water quality status of the Lake Nubia is excellent, based on the measured parameters, and good, based on the simulated parameters. AME is 4.25. This is because two of the simulated values of one parameter, pH, are out of the objective range, while all measured water quality parameters values are in the objective range.

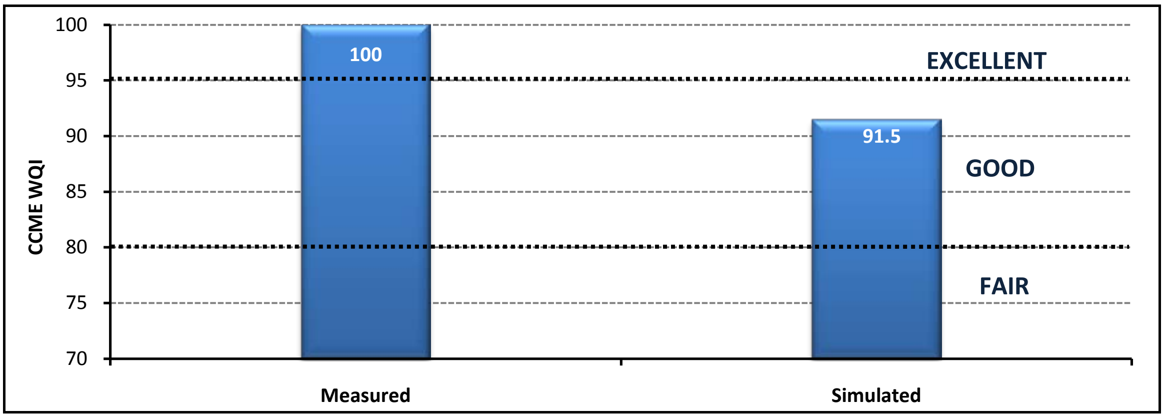

Figure 5.5: Lake Nubia CCME WQI for measured and simulated water quality parameters, January 2006.

5.3.3.3 Carlson TSI

The Carlson TSI was used to evaluate the trophic state of Lake Nubia. Two water quality parameters were used to estimate the Carlson TSI: total phosphorus (µg/l) and chlorophyll-a (µg/l). The Secchi depth was excluded because the lake transparency is affected by suspended particles rather than by algae; the southern part of AHD reservoir, Lake Nubia, has a high total suspended solids concentration.

Figure 5.6 shows the Carlson TSI longitudinal profiles of the Lake Nubia for measured and simulated total phosphorus, January 2006. The results show that according to Carlson TSI, based on the total phosphorus parameter, the trophic state of the Lake Nubia is hypereutrophic. The Carlson TSI varies spatially; it decreases from 88 (for measured data) at St. 2 to 80 at St. 9. This corresponds with the total phosphorus decrease due to algae activity.

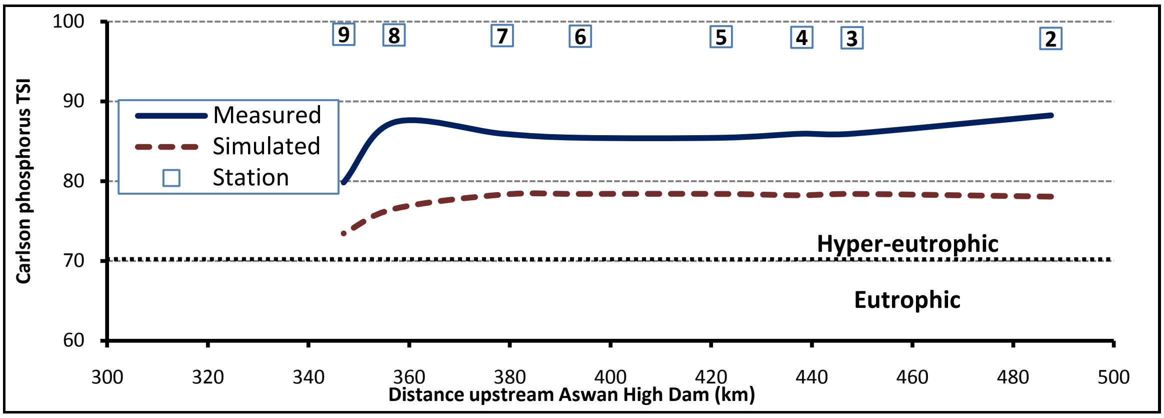

Figure 5.6: Lake Nubia Carlson TSI longitudinal profiles for measured and simulated total phosphorus parameter, January 2006.

Lake Nubia Carlson TSI longitudinal profiles for measured and simulated chlorophyll-a parameter, January 2006, are shown in Figure 5.7. The results show that Lake Nubia trophic state, according to Carlson TSI (based on chlorophyll-a parameter), is almost eutrophic. The index slightly varies along Lake Nubia till St. 8 where it starts to increase due to algae activity.

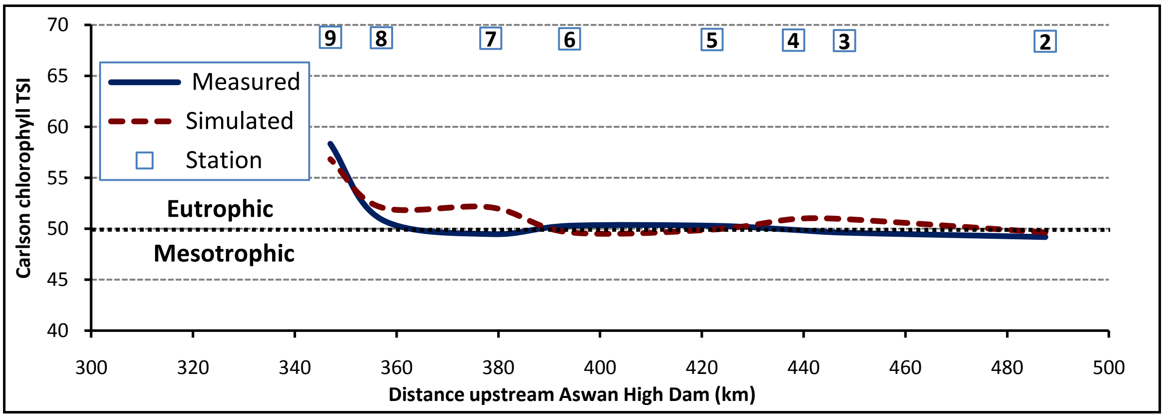

Figure 5.7: Lake Nubia Carlson TSI longitudinal profiles for measured and simulated chlorophyll-a parameter, January 2006.

The difference between Carlson TSI, based on total phosphorus, and that one based on chlorophyll-a may return to phosphorus surplus in the water column (Abou El Kheir et al., 2003).

Figures 5.8 and 5.9 show the average Carlson TSI longitudinal profiles of the Lake Nubia for the measured and simulated water quality parameters in January 2006. According to Carlson TSI, the Lake Nubia trophic state is eutrophic. The simulated results closely match the index result based on measured parameters; the AME is 1.9.

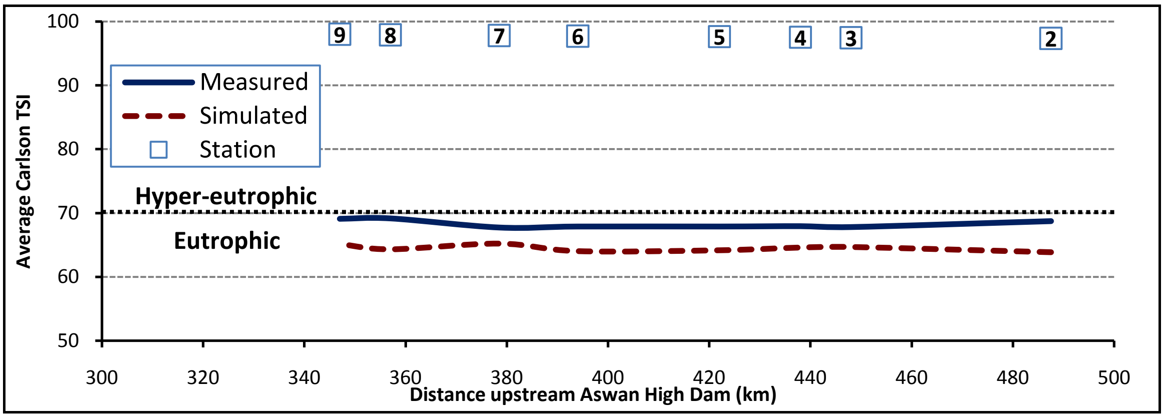

Figure 5.8: Lake Nubia average Carlson TSI longitudinal profiles for measured and simulated water quality parameters, January 2006.

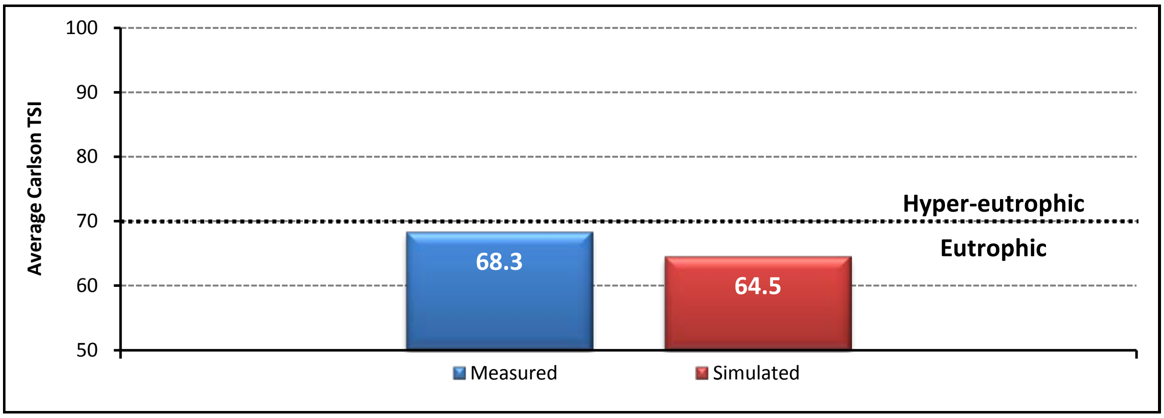

Figure 5.9: Lake Nubia average Carlson TSI for measured and simulated water quality parameters, January 2006.

5.3.3.4 LAWA TI

The second index which has been used to evaluate the trophic state of Lake Nubia is LAWA TI. As in Carlson TSI, Two water quality parameters were used; total phosphorus (µg/l) and chlorophyll-a (µg/l). Secchi depth was excluded because transparency is affected by suspended particles rather than by algae.

Figure 5.10 shows the Lake Nubia LAWA TI longitudinal profiles for measured and simulated parameters, January 2006. The results show that the Lake Nubia trophic state is eutrophic. The index slightly varies spatially till St. 8 where it starts to increase due to algae activity.

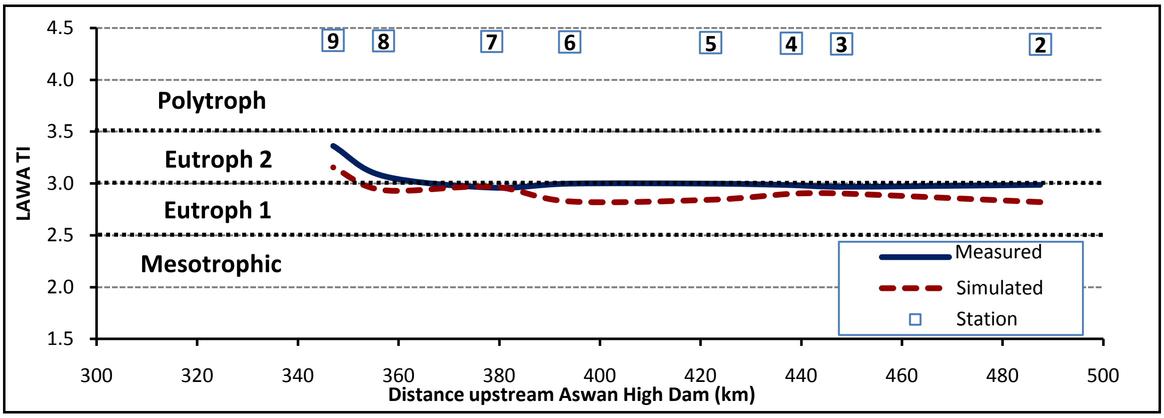

Figure 5.10: Lake Nubia LAWA TI longitudinal profiles for measured and simulated water quality parameters, January 2006.

The Lake Nubia average LAWA TI for measured and simulated water quality parameters, January 2006, is shown in Figure 5.11. The simulated results closely match the index result based on the measured parameters; the AME is 0.06. According to LAWA TI, the Lake Nubia trophic state is eutrophic.

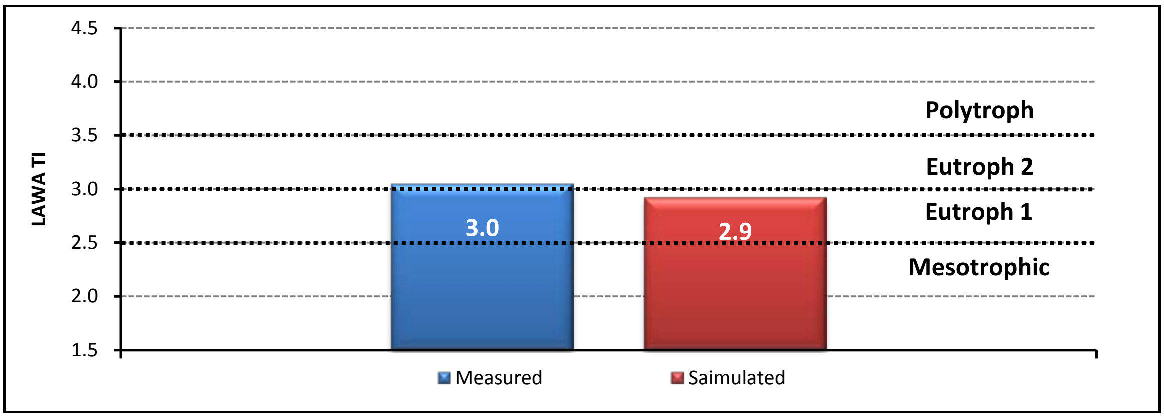

Figure 5.11: Lake Nubia average LAWA TI for measured and simulated water quality parameters, January 2006.

5.3.4 Results summary

Table 5.7 summarizes the results of the application of the four water quality and trophic status indices for Lake Nubia in January 2006, based on the measured and simulated water quality characteristics. Moreover, Table 5.7 includes the average AME for each index.

Table 5.7: A summary of WQI and TSI study results for Lake Nubia in January 2006.

	Water quality Indices (WQIs)		Trophic status Indices (TSIs)	
	NSF WQI	CCME WQI	Carlson TSI	LAWA TI
Measured parameters	Good	Excellent	Eutrophic	Eutroph 1
Simulated parameters	Good	Good	Eutrophic	Eutroph 1
Average AME	0.4	4.25	1.9	0.06

5.3.5 Conclusions

Two water quality indices, NSF WQI and CCME WQI, were developed to assess water quality state of the southern part of AHD reservoir, Lake Nubia, during low flood period, January 2006. Moreover, another two trophic status indices, Carlson TSI and LAWA TI, were developed to evaluate trophic status of Lake Nubia during the same period, January 2006. Results of a specially developed hydrodynamic and water quality model for Lake Nubia have been used to check the model validity to simulate the reservoir.

1. According to the developed water quality indices results, Lake Nubia has a good water quality state during the low flood period. The modified CCME WQI, based on the measured data, indicates that the Lake Nubia water quality state is excellent, according to the Egyptian water quality standards for surface waterways.
2. Results of the applied trophic status indices show that the Lake Nubia trophic status is eutrophic. The Carlson TSI, based on total phosphorus, indicates that the trophic status of Lake Nubia is hypereutrophic.
3. The proposed hydrodynamic and water quality model shows a good agreement between the indices estimation based on the measured and the simulated parameters.

4. The morphological characteristics of Lake Nubia affect water quality and trophic states of the reservoir; the transition zone of the reservoir (starts from St. 7) has a better water quality state and a little bit worse trophic state than the riverine zone (upstream St.7). This indicates that reservoir zones should be assigned to different water uses according to its water quality and trophic states.
5. For the AHD reservoir, Secchi depths should not be used to estimate the trophic status of the reservoir especially in the riverine and transitional zones where the water transparency is affected by suspended particles rather than by algae.

5.3.6 Recommendations for future work

1. A scientific study of the long term trend of the water quality and trophic status should be done for the AHD reservoir using a detailed database. Such a study could be the basis for a control management of the reservoir water quality.
2. Egyptian water quality standards for different uses (e.g.: irrigation water, fishing, swimming) should be developed and used as guidelines in different water quality indices.
3. The NSF WQI should be developed according to different water uses instead of the general water quality state.
4. The CCME WQI procedure should have more guidelines about parameters choice and a more detailed ranking for the water quality state especially if all variables do not exceed the guidelines.
5. For trophic status indices, more than one parameter should be used. Moreover, a newer index should be developed for sub-tropical lakes and reservoirs such as the AHD reservoir.
6. As the trophic status is an aspect of water quality, a detailed study should be done to investigate the relation between water quality and trophic status indices.

6. Global climate change impacts

The global climate change (GCC) due to the greenhouse effect is now firmly on the world's scientific and political agenda. Observational evidence from all continents and most oceans shows that many natural systems are being affected by regional climate changes, particularly temperature increases (IPCC, 2007a). In this chapter, the present status of the climate change theory and its effects on water resources characteristics will be briefly reviewed. Moreover, climate change impacts on Lake Nubia with respect to the hydrodynamics and the water quality will be investigated.

6.1 Climate change state-of-the-art

Climate change is a “normal” climatic phenomenon; the climate has always been changing throughout the history of the Earth. Natural changes in climatic conditions have resulted in Ice Ages and relatively warm periods in temperate regions while wet periods have intermitted with dry periods in Africa (Van der Geest, 2002). Climate change, since the last century, however has been accelerated due to the increase in greenhouse gases related to human activities (CECC, 2008).

In 1896 Svante Arrhenius, the Nobel prize awarded Swedish scientist, made a remarkable prediction, warning that if coal burning were to double the concentration of CO_2 in the atmosphere, the temperature of the earth could rise by “several degrees”. Three-quarters of a century later, in 1972, J. S. Sawyer of the U.K. Meteorological Office predicted that the CO_2 increase of 25%, by the end of 20th century, would correspond to a 0.6°C rise in world temperature (Svendsen and Künkel, 2008). The first world climate conference was held in 1979 to urge the nations of the world to improve their knowledge about climate and prevent man-made changes in climate (Seki and Christ, 1995). The Intergovernmental Panel on Climate Change (IPCC) was established in 1988 by the United Nations Environment Programme and the World Meteorological Organization to assess the environmental and socioeconomic implications of climate change and the possible response options available to governments. IPCC has released four assessment reports in 1990, 1995, 2001 and 2007.

The latest IPCC Assessment Report (IPCC, 2007a; b; c) stated that Earth’s average temperature is unequivocally warming, 11 of the past 12 years have been the warmest on record since 1850, which is when global instrumental record-keeping began. The report documented that anthropogenic factors (due to human activity) are responsible for most of the current global warming. The primary anthropogenic source is the emission of greenhouse gases such as carbon dioxide, which is mainly produced by the burning of fossil fuels.

Greenhouse gases are a natural part of the atmosphere, without these gases the global average temperature would be around -20 °C (UNEP and GRID-Arendal, 2005). Solar radiation is absorbed by the Earth’s surface and it later leaves the Earth as infrared radiation. However, part of the outgoing infrared radiation is absorbed by greenhouse gases in the atmosphere and re-emitted to the Earth’s surface, which makes the Earth warmer. The more of these gases there are the more heat is trapped, this is known as the enhanced greenhouse effect (UNEP and GRID-Arendal, 2005). Global greenhouse gas (GHG) emissions have grown since pre-industrial times, with an increase of 70% between 1970 and 2004 (IPCC, 2007b). Figure 6.1 shows greenhouse gases mechanism.

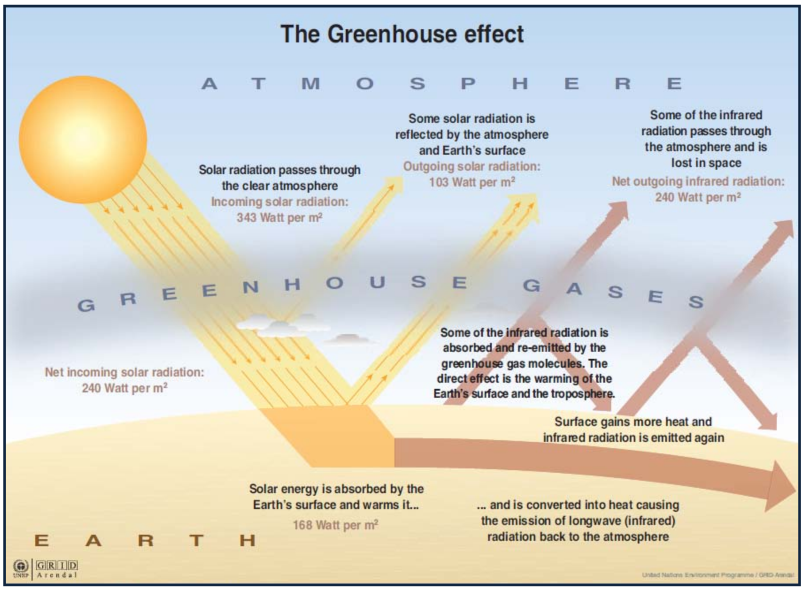

Figure 6.1: Greenhouse gases mechanism (UNEP and GRID-Arendal, 2005).

Although scientists are confident about the fact of global warming and climate change due to human activities, substantial uncertainty remains about just how large the warming will be and what will be the patterns of change in different parts of the world (Houghton, 2004). Different global climate models or General Circulation Models (GCMs) were designed by various groups of scientists and used to predict the impact of enhanced greenhouse effect on climate change, by using different emission scenarios. A wide range of emission scenarios was developed by the IPCC in a Special Report on Emission Scenarios, which is available at web site of (SRES website). The main scenarios storylines are as follows (Storch, 2008):

- A1 Storyline describes a world of rapid economic growth and rapid introduction of new and more efficient technology.
- A2 Storyline describes a very heterogeneous world with an emphasis on family values and local traditions.
- B1 Storyline describes a world of "dematerialization" and introduction of clean technologies.
- B2 Storyline describes a world with an emphasis on local solutions to economic and environmental sustainability.

IPCC projects that global average temperatures in 2100 will be between 1.8–4.0 °C higher than the 1980–2000 average (best estimate, likely range 1.1–6.4 °C). Sea levels are projected to rise 0.18–0.59 m by 2100. More frequent and intense extreme weather events (including drought and flooding) are also expected. Figure 6.2 shows temperature projections to the year 2100, based on a range of emission scenarios and global climate models. The orange line of "Constant

CO_2" projects global temperatures with greenhouse gas concentrations stabilized at year 2000 levels.

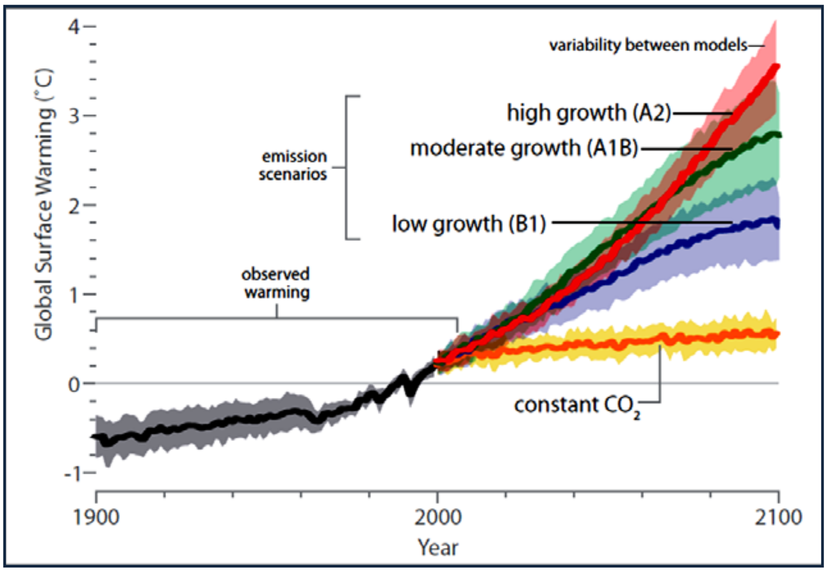

Figure 6.2: Warming projections to the year 2100 (USEPA, 2009)

6.2 Climate change impacts on water resources

The impacts of climate change on freshwater systems and their management are mainly due to the observed and projected increases in temperature, sea level and precipitation variability (very high confidence), Semi-arid and arid areas are particularly exposed to the impacts of climate change on freshwater (high confidence) (Kundzewicz et al., 2007). Observations indicate that lakes and rivers around the world are warming, with effects on thermal structure and lake chemistry that in turn affect abundance and productivity, community composition, phenology, distribution and migration (Rosenzweig et al., 2007). Higher water temperatures, increased precipitation intensity and longer periods of low flows are projected to exacerbate many forms of water pollution, including sediments, nutrients, dissolved organic carbon, pathogens, pesticides, salt and thermal pollution. This will promote algal blooms, and increase the bacterial and fungal content. This will, in turn, impact ecosystems, human health and the reliability and operating costs of water systems (Bates et al., 2008). European Environment Agency (EEA, 2007) has listed some expected water quality problems for water resources because of temperature changes as follows:

- Reduced oxygen content. Increases in water temperature in streams and rivers reduce oxygen content and increase biological respiration rates and thus may result in lower dissolved oxygen concentrations, particularly in summer low-flow periods.
- Alterations to habitats and distribution of aquatic organisms.
- Alterations to thermal stratification and mixing of water in lakes.
- Changed nutrient cycling in aquatic systems and algal blooms.
- Increase of bacterial populations that control nitrogen mineralization and nitrification processes in the soils.

Detailed expected climate change impacts on water resources have been discussed in numerous studies (Bates et al., 2008; Gleick, 1987; Gleick, 1998; Grimm et al., 1997; Kundzewicz et al., 2007; Mulholland et al., 1997; Murdoch et al., 2000; Schindler, 1997; Whitehead et al., 2009). State-of-the-art research into the implications of climatic change for the hydrologic cycle and for water resources has been reviewed in some other publications (Bates et al., 2008; Delpla et al., 2009; Gleick, 1989; Leavesley, 1994; Varis et al., 2004; Whitehead et al., 2009).

GCMs and hydrological models have been used to predict future hydrological characteristics of rivers basins such as runoff (Block et al., 2009; Christensen et al., 2004; Gleick, 1987; Middelkoop et al., 2001). The effects of climate change on the discharge regime in different parts of the Rhine basin were calculated using the results of UKHI and XCCC GCM-experiments (Middelkoop et al., 2001). Different detailed hydrological models with hourly and daily time steps have been developed. All models indicate the same trends in the changes: higher winter discharge as a result of intensified snow-melt and increased winter precipitation, and lower summer discharge due to the reduced winter snow storage and an increase of evapotranspiration.

Hondozo and Stefan (1993) addressed three approaches can be used to investigate climate change impacts on lakes water temperature: examination long-term records, comparing individual warm and cold or wet and dry years if the records are short but detailed, or using numerical simulation models to calculate heat transfer from the atmosphere to the lake water columns.

Studies of climate change impacts based on past long-term records have been addressed in the literature (Coats et al., 2006; Fukushima et al., 2000; George et al., 2007; Koinig et al., 1998; Livingstone, 2003; O'Reilly et al., 2003; Ozaki et al., 2003; Sahoo and Schladow, 2008; Troin et al., 2008). In 2000, the effects of past long-term, 1979-1996, climate change on the water quality of Lake Kasumigaura, Japan, were investigated (Fukushima et al., 2000). The authors developed a statistical regression relationships between the past meteorological conditions and lake water quality parameters. They reported that the deterioration of lake water quality, such as increases in COD and decreases in transparency, was quantitatively assessed as corresponding to an increase in air temperature. In addition, they found that higher precipitation led to high nitrogen concentrations on a monthly basis, as well as on a yearly basis. O'Reilly et al. (2003) have investigated the effect of past climate change on water temperature and the water column stability, which is defined as the work required to mix the water column to uniform density, of Lake Tanganyika, Africa. The authors stated that in parallel with regional warming patterns since the beginning of the twentieth century, a rise in surface-water temperature has increased the stability of the water column. A regional decrease in wind velocity has contributed to reduced mixing, decreasing deep-water nutrient upwelling and entrainment into surface waters. In 2008, The effects of past (1969-2002) and future (2000-2040) climate change on the physical characteristics of Lake Tahoe, California – Nevada, USA, were investigated (Sahoo and Schladow, 2008). For future climate change study, one emission scenario (A2) for one GCM (GFDL) and two models (watershed model and hydrodynamic and water quality model), were used. The past records show that Lake Tahoe has become warmer and more stable, while the future predicted trends show that the lake continues to become warmer and more stable, and mixing is reduced.

The authors discussed the possible changes in water quality because of global warming, they expect that if the warming trend continues, lakes will be permanently stratified, resulting in low DO concentrations in the hypolimnion.

Climate change impacts on the hydrodynamic characteristics of water resources have been investigated in numerous publications (Anderson et al., 2008; Fang et al., 2007; Fang and Stefan, 1999; George et al., 2007; Gooseff et al., 2005; Hassan et al., 1998a; Hondzo and Stefan, 1993; Qin and Huang, 1998; Sahoo and Schladow, 2008; Sinokrot et al., 1995; Stefan et al., 1998). A validated, one-dimensional, unsteady lake water quality model was used to simulate 27 classes of lakes in Minnesota, USA, for past daily time base (1955-1979) and future climate scenario (Hondzo and Stefan, 1993). One emission scenario was used; double CO_2 for one GCM (GISS). The simulations predict that epilimnetic temperatures will be higher, hypolimnetic temperatures in seasonally stratified dimictic lakes will be largely unchanged or even lower than at present, evaporative water loss will be increased by as much as 300 mm for the season, onset of stratification will occur earlier and overturn later in the season, and overall lake stability will become greater in spring and summer. Similar studies have been done later for the same 27 classes of lakes with different water quality model and GCM (Fang and Stefan, 1999; Stefan et al., 1998). In 1998, climate change effects o the hydrodynamic characteristics of Lake Qinghai, Qinghai-Tibet Plateau, China, have been investigated (Qin and Huang, 1998). Catchment model, lake thermodynamic model and lake water balance model were developed. The results of 4 GCMs (GFDL, GISS, OSU and UKMO) for doubling CO_2 were used. The results show that the total runoff in the lake and evaporation will, in most cases, increase as conditions become warmer and wetter. The lake level changes would remain uncertain because the effects of an increase in precipitation are countered by the rise of temperature. The impact of future climate change on the thermal structure of Amistad reservoir, Texas, USA, was examined (Fang et al., 2007). CE-QUAL-W2, the hydrodynamic and water quality model, was developed and the result of one GCM, CCC GCM, for doubling CO_2 was used. The results show a strong impact on both surface and bottom water temperature in the reservoir. The stratification of temperature is projected to last for a longer period, but its strength is reduced.

Up to now, the effects of climate change on the water quality of water resources have been investigated in limited publications only (Chang et al., 1992; Fanga et al., 2004a; b; c; Hassan et al., 1998b; Hosomi et al., 1996; Komatsu et al., 2007; Malmaeus et al., 2006). One of the first trials to study the impact of climate change on fish habitat in Douglas reservoir, Tennessee (USA), was done in 1992 (Chang et al., 1992). The authors used the results of three GCMs (two grid cells for each) for double CO_2 and a water quality model. A lot of assumptions were used to estimate most of the required input data of the water quality model. Daily reservoir volumes with optimal, suboptimal, and unsuitable temperature and dissolved oxygen were predicted for the year. The authors conclude that the reservoir model was found to be a promising tool for examining potential climate-change impacts. However, some of the assumptions required to apply GCM output to the reservoir model illustrate the problems of using large-scale grid cell output to assess small-scale impacts.

In 1996, a new water temperature-ecological model was applied to Lake Yunoko, Japan (Hosomi et al., 1996). The calibrated model was used to assess the effects of global warming by increasing air temperature to 24 °C. Significant impacts on thermal, chemical and biological characteristics of the lake were found. In 1998, a new mathematical eutrophication model to simulate phytoplankton growth rate and dissolved oxygen for Suwa Lake, Japan, was developed (Hassan et al., 1998b). The authors assessed the impact of future climate change on phytoplankton growth rate, a downscaling method was applied to (HadCM2SUL) GCM for (2080-2099).

In 2004, a study of three parts (Fanga et al., 2004a; b; c) was done to assess the impact of climate change on fish habitat, which is strongly constrained by water temperature and available dissolved oxygen (DO). The authors used the same water quality model, emission scenario and GCM for the same 27 classes of lakes of that study of Fang and Stefan (1999). The vertical DO profiles in the lake are computed from a balance between oxygen sources (reaeration and photosynthesis) and oxygen sinks (sedimentary oxygen demand, biochemical oxygen demand and plant respiration). The authors reported that errors between simulated and measured DO concentrations are in part caused by using constant biochemical oxygen demand (BOD) and sediment oxygen demand (SOD) regardless of season and location, although they depend on the trophic status and lake depth in the model.

In 2006, the climate warming impact on phosphorus dynamics in lakes was investigated (Malmaeus et al., 2006). A physical lake model and a mechanistic phosphorus model were combined with two temperature scenarios generated by a regional climate model (RCM) in three sites in central Sweden—Lake Erken and two basins of Lake Mälaren. The model results indicated that lakes may respond very differently to climate change depending on their physical characteristics, especially water residence time. In Lake Erken the concentration of epilimnetic-dissolved phosphorus is almost doubled in spring and autumn in the warmest climate scenario, since the lake is mostly phosphorus limited, this means that the potential for phytoplankton production is almost doubled. In other eutrophic lakes with long water residence times, eutrophication problems may become serious in the future.

The long-term effect of global warming on environmental variables, such as water temperature, dissolved oxygen and nutrients as well as aquatic ecosystems was evaluated by Komatsu et al. (2007). The developed watershed runoff model and reservoir water quality model with meteorological input calculated by a GCM A2 scenario was applied to Shimajigawa reservoir, Japan. The results were shown to cause more trophic lake conditions, further promoting algal growth and changing the aquatic ecosystems.

6.3 Climate change effects on the Egyptian water resources

Developing countries, such as Egypt, will be strongly threatened by the hydrological impacts of global climate change (GCC). This is because many of the poorest countries lie in those regions where GCC-related effects will be most damaging, and their ability to respond to harmful change is extremely limited (Svendsen and Künkel, 2008). The major, and sometimes the only, viable option for developing countries is adaptation. The strategies developed under the National

Water Resources Plan (NWRP) are categorized into three main directions: optimal use of available resources, development of new resources and water quality preservation/improvement (Attia, 2008). The first and second Egyptian communications reports (EEAA, 1999; 2010) which were prepared by the Egyptian Environmental Affairs Agency (EEAA) to submit to the United Nations Framework Convention on Climate Change (UNFCCC) reported the following facts: Egypt is one of the most vulnerable countries to the potential impacts and risks of climate change, even though it produces less than 1% of the world total emissions of GHG, with a vulnerability of all sectors of development and a low resilience of the majority of stakeholders. The sectors of water resources, agricultural resources and food security, coastal resources, tourism and health are highly vulnerable with serious socioeconomic implications.

More than 95% of the water budget of Egypt is received from the River Nile which is generated outside Egypt's territory. Numerous studies showed that River Nile is very sensitive to temperature and precipitation changes (Riebsame et al., 1995) mainly because of its low runoff/rainfall ratio (4%). Therefore, it is of prime importance for Egypt, amongst other Nile countries, to assess the hydrological impacts of climate change on the River Nile (Attia, 2008). The implications of climate fluctuations for water management with emphasis on the Nile were considered in some publications (Abu-Zeid and Biswas, 1991; Conway and Hulme, 1993). A lot of studies have investigated the potential impacts of climate change on the water resources of the River Nile and associated impacts on the Egyptian economy (Jeuland and Whittington, 2008; Strzepek and Yates, 2000; Strzepek et al., 2001; Yates and Strzepek, 1998; Yates and Strzepek, 1996). Moreover, the impacts of climate change on the Egyptian agriculture sector have been detailed in numerous publications (Abou Hadid, 2006; Abou Hadid and Eid, 2006; El-Shaer et al., 1997; Mougou et al., 2008).

Gleick (1991) analyzed the vulnerability of Nile runoff to climate change with the help of a sensitivity study. He applied a theoretical model (based on the annual water balance) to three sub-basins of the Nile Basin, the Upper White Nile, Sobat, Blue Nile and Atbara. The model produced a 50% reduction in runoff in the Blue Nile catchment due to an assumed 20% decrease in precipitation.

In 1995 three global climate models (GCMs) for doubled CO_2 levels and a hydrological model to generate multi-year time series of monthly stream flow at Aswan were used (Riebsame et al., 1995). The GCMs simulations showed an increase of 30% and two decreases, with one of the decreases greater than 70% of the annual flow.

The application of hydrologic models of the Blue Nile and Lake Victoria sub-basins to assess the magnitude of potential impacts of climate change on Main Nile discharge was described in 1996 (Conway and Hulme, 1996). The models are calibrated to simulate historical observed runoff and then driven with the temperature and precipitation changes from three general circulation model (GCM) climate scenarios; a "wet" case (GISS GCM), "dry" case (GFDL GCM) and composite case (a weighted mean of seven equilibrium GCMs). The following changes in Egypt's allocation of Nile water (estimated as about 55 km^3), due to 1 °C temperature increase, were obtained: +0.8 km^3 (-3.6 km^3), -4.9 km^3 (-2.6 km^3) and +8.4 km^3 (+2.6 km^3) with both temperature and precipitation changes (temperature only) applied from the composite, GFDL and GISS GCM scenarios, respectively.

In 1996, the impacts of global climate change on the water resources of the River Nile Basin were evaluated using four climate change scenarios (baseline, GISS, GFDL and UKMO) (Strzepek et al., 1996). The authors concluded that the complete impact of climatic changes on the Nile cannot be fully predicted with confidence, because some models forecast increased flows, while others project significant decreases. The River Nile flows under GCMs scenarios were as follows: 88% (UKMO GCM), 130% (GISS GCM) and 23% (GFDL GCM). The authors concluded that it is possible that the effects of climatic fluctuations on the River Nile would be severe.

The Organization for Economic Cooperation and Development (OECD) has published a study which analyzes the climate change significant risks on Egypt (Agrawala et al., 2004). Changes in area averaged temperature and precipitation over Egypt were assessed based upon the results of 8 GCMs for IPCC B2 emission scenario. The study also examined the climate models projections for the source waters of the Nile, in the Ethiopian highlands and equatorial lakes region. Temperature changes there are expected to be similar, but precipitation changes are vary.

Conway (2005) refers that studies since the early 1990s highlight the low convergence of climate model scenarios of rainfall over the Nile basin while climate scenarios of rising temperatures are more consistent and could lead to large increases in the evaporation because of the large expanses of open water and irrigated agriculture in the basin.

A regional climate model has been applied to the Nile Basin (Mohamed et al., 2005). The model has been customized to simulate the regional climate of the Nile (tropical, semi arid and arid climates). The exercise concentrated on reproducing the regional water cycle as accurately as possible. Observations on runoff, precipitation, evaporation and radiation have been used to evaluate the model results at the sub-basin level (White Nile, Blue Nile, Atbara and the Main Nile). Subsequently, the model has been used to compute the regional water cycle over the Nile Basin. Another regional climate model has been calibrated by using different observed data for Blue Nile and Sobat River sub-basins (Soliman et al., 2008). The results of this regional climate model were used to simulate the runoff pattern by using a hydrological model which is connected to the regional climate mode.

Sayed (2008) has reported some needed research points which help in studying climate change risks on the Nile basin such as calibration and validation of a regional climate model to test the impacts of extreme scenarios on the spatial and temporal distribution of rainfall over the Nile Basin.

Attia (2008) has assessed and analyzed the Egyptian water resources system vulnerability under climate change by presenting the results of climatic scenarios based on three GCMs (CGCM2, ECHAM4 and HadCM3) for two emission scenarios (A2 and B2). These three models were selected based on the Lake Nasser Flood and Drought Control project (LNDFC) which constructed climatic scenarios (low, central, high) based on the results from 11 GCMs for the SRES B2 emission scenario. These scenarios were taken from the OECD study (Agrawala et al., 2004). The study used these uniform changes to estimate the impacts of climate change on the inflows to Lake Nasser showing a wide range of changes for 2030, 2050 and 2100. According to the low scenario, Egypt's share of the Nile waters declines to only 7 BCM by 2100, rendering irrigated agriculture impossible. On the contrary, the high scenario would permit doubling the irrigated area by 2100. The central scenario predicts a modest increase in the annual share of about 3, 5 and 8 BCM by 2030, 2050 and 2100 respectively.

In 2009, the impacts of climate change on both hydrology and water resources operations of Blue Nile River were analyzed using the outcomes of six different general circulation models (GCMs) for the 2050s (Kim and Kaluarachchi, 2009). The changes in outflows from two proposed dams to downstream countries were also estimated. Given the uncertainty of different GCMs, the simulation results of the weighted scenario suggested mild increases in hydrologic variables (precipitation, temperature, potential evapotranspiration and runoff) across the studied area.

The outputs of 17 GCMs for A1B emission scenario, included in the 4th IPCC assessment report, were downscaled for the 2081– 2098 period for the upper Blue Nile basin (Elshamy et al., 2009b). These were used to drive a fine-scale hydrological model of the Nile Basin to assess their impacts on the flows of the upper Blue Nile at Diem, which accounts for about 60% of the mean annual discharge of the Nile at Dongola. The ensemble mean annual flow at Diem is reduced by 15% compared to the baseline. However, the very high sensitivity of flow to rainfall makes such a number highly uncertain.

Elshamy et al. (2009a) analyzed the change in Nile flows at Dongola using the results of 3 GCMs (CGCM2, ECHAM4 and HadCM3) for two emission scenarios (A2 and B2). The GCMs results were downscaled and bias corrected by using a statistical downscaling model. A fine-scale hydrological model of the Nile basin was used to simulate the inflow. The results showed that the ECHAM4 predicts a steady increase in Nile flows in the future. CGCM2 underestimates the flow compared to the mean observed flow during the base period (1992-2001). The HadCM3 showed a slight trend of increase to the flood season flows and a slight trend of increase for the flow during the base-period.

A simplified model was used to simulate two different future scenarios to study the effect of the water temperature rise (in the Rosetta branch of River Nile) on the DO concentrations (Radwan, 2009). The effect of climate change has been investigated by assuming air water temperature rise by 1.5 °C and 3 °C and another reduction factor has been applied to have the water temperature rises according to the air temperature increase. The results indicated that Global warming might increase the stress on Rosetta Branch and decline its water quality. If this effect was accompanied by flow reduction, deterioration in the quality status of the branch might be severe.

The evaporation losses from Aswan High Dam Reservoir (AHDR) at 2050 due to the expected climate changes have been evaluated (Badawy, 2009). The author used two different meteorological data sets to obtain the different meteorological parameters trend. He applied seven methods to calculate evaporation losses for the present and in the future. The obtained results showed that the evaporation losses from AHDR will be increased by a very negligible change.

In 2010, the potential impacts of climate change on the hydrology of the River Nile basin for three periods in the 21st Century, period I (2010-2039); period II (2040-2069); and period III (2070-2099), have been assessed (Beyene et al., 2010). The authors used the results of eleven GCMs for two global emissions scenarios (A2 and B1), archived from the 2007 IPCC Fourth Assessment Report (AR4). The archived results were bias corrected and spatially and temporally downscaled, by disaggregating the values of temperature and precipitation at the GCM spatial scale in space (to the 1/2° latitude–longitude resolution) and in time (from monthly to daily). These results were used as the input data of a macro-scale hydrological model to predict the river inflow. The simulations show that, averaged over all 11 GCMs, the River Nile is expected to experience increase in stream-flow early in the study period (2010–2039), due to generally increased precipitation. Stream-flow is expected to decline during mid- (2040–2069) and late (2070–2099) of the century as a result of both precipitation declines and increased evaporative demand. The predicted multi-model average stream-flows at Aswan High Dam (AHD) as a percentage of historical (1950– 1999) annual average are 111 (114), 92 (93) and 84 (87) for A2 (B1) global emissions scenarios. Implications of these stream-flow changes on the water resources of the River Nile basin were analyzed by quantifying the annual hydropower production and irrigation water release at AHD.

6.4 Research deficits and demands for this study

As addressed before, see section 6.2, up to now, the effects of climate change on the water quality of water resources have been investigated in limited publications only using a lot of assumptions. Here in this research work, the climate change impacts on hydrodynamic and water quality characteristics of the investigated area were investigated. The future initial conditions of water temperature were estimated by using the historical measured data base of the investigated area for different years. A theoretical process algorithm was simplified, further developed and calibrated to estimate the future initial conditions of DO due to global climate change effects.

Egypt is one of the most vulnerable countries to the potential impacts and risks of climate change, see the previous section. More than 95% of the freshwater budget of Egypt is received by the River Nile which is very sensitive to climatic changes. The impacts of climate change on the River Nile flow have been investigated in numerous studies, while the effects of climate change on the hydrodynamic or water quality characteristics of the different Egyptian water resources were rarely investigated. In this research work, the impacts of climate change on the hydrodynamic and water quality characteristics of the investigated area were investigated using the predictions of eleven GCMs (for two emission scenarios) for the 21st century.

6.5 Case study: Lake Nubia

In the following, the climate change impacts on hydrodynamic and water quality characteristics of Lake Nubia are investigated. Moreover, the climate change effects on water quality and trophic status indices are quantified.

6.5.1 Climate change estimates

In order to investigate the impact of future climate change scenarios on the hydrodynamic and water quality characteristics of Lake Nubia, projected changes in climate conditions are required. These conditions have been obtained from the study of Beyene et al. (2010) with reference to section 6.3 for more details. Figure 6.3 summarizes the used approach. Figure 6.4 shows the average annual temperature (°C) and precipitation (%) Changes (relative to 1950-1999 historical average), for each one of 11 GCMs, two global emission scenarios (A2 and B1) and three periods: I (2010-2039), II (2040-2069) and III (2070-2099). The downscaled bias corrected air temperature changes, precipitation changes and simulated main Nile stream flow changes for all scenarios are represented in Figures 6.5 – 6.7. These data are used in this work to modify the input files of the proposed CE-QUAL-W2 hydrodynamic and water quality model of Lake Nubia.

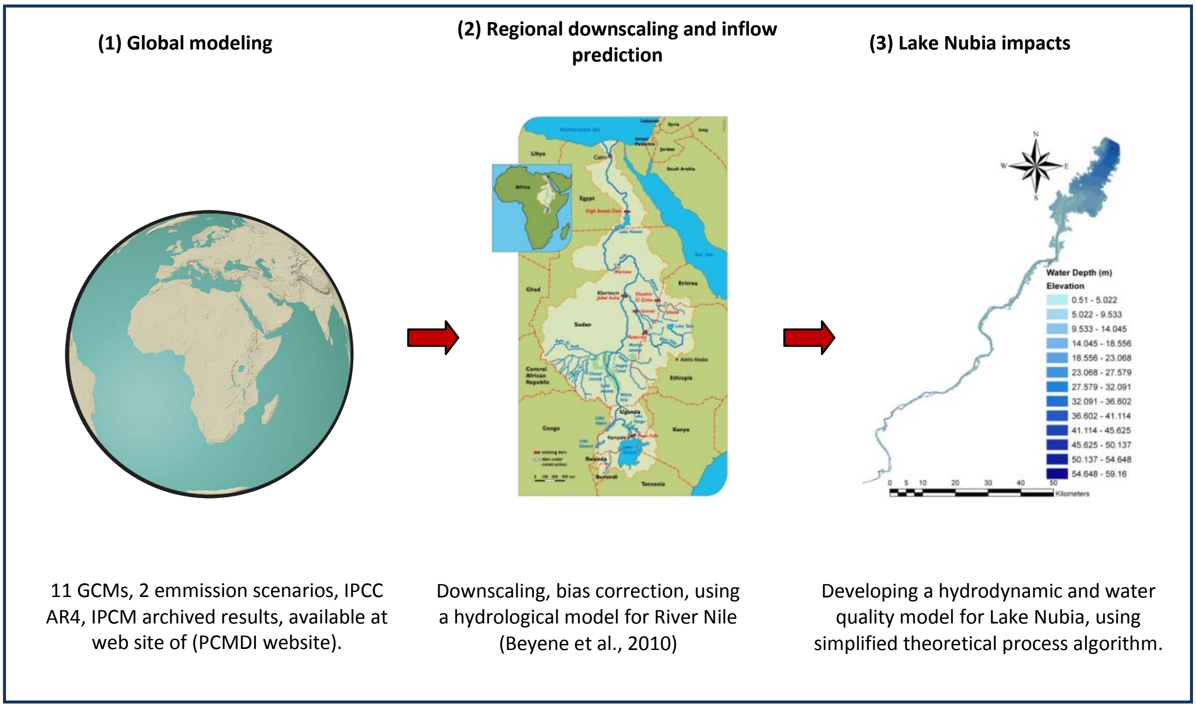

Figure 6.3: Approach for analyzing potential impacts of climate change on Lake Nubia; step (3) is performed by the author.

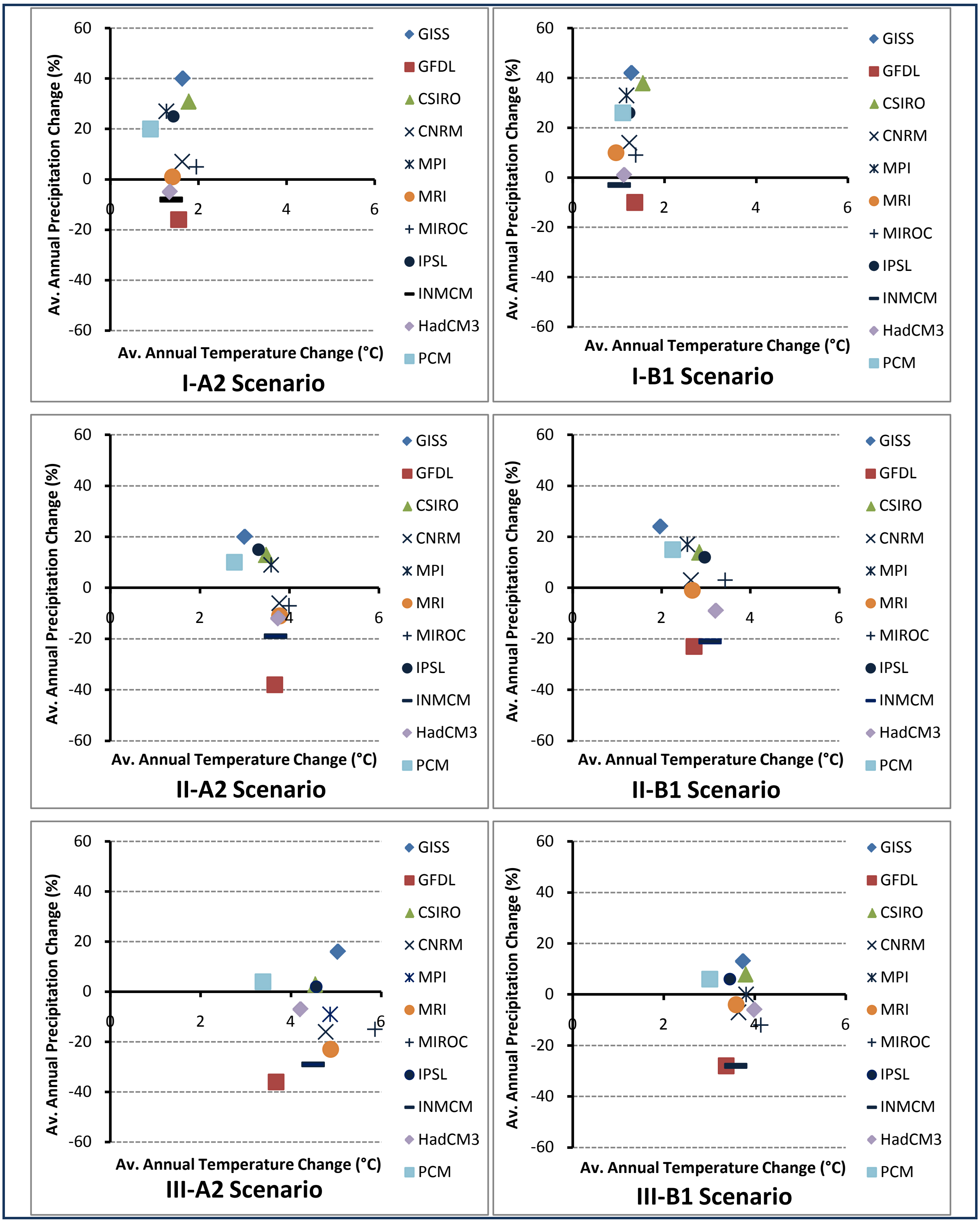

Figure 6.4: Average annual temperature (°C) and precipitation (%) changes (relative to 1950-1999 historical average), for each one of 11 GCMs, two global emission scenarios (A2 and B1) and three periods: I (2010-2039), II (2040-2069) and III (2070-2099).

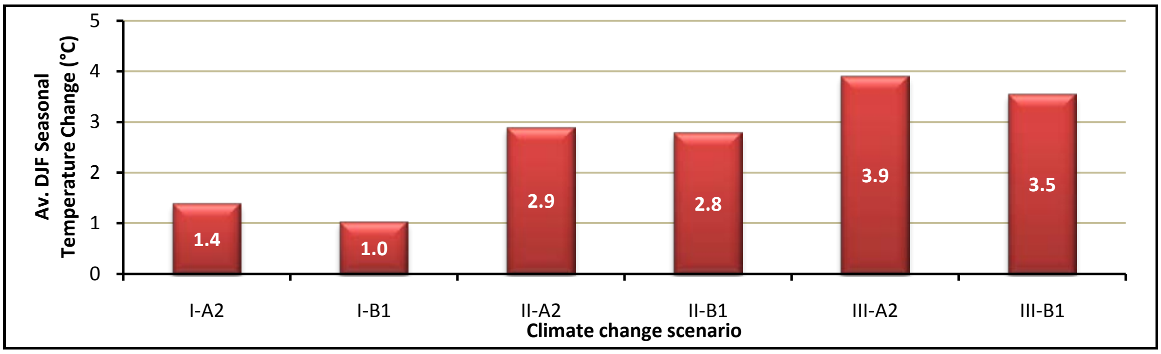

Figure 6.5: Average multi-model DJF (December, January and February) seasonal temperature changes (°C) (relative to 1950-1999 historical average).

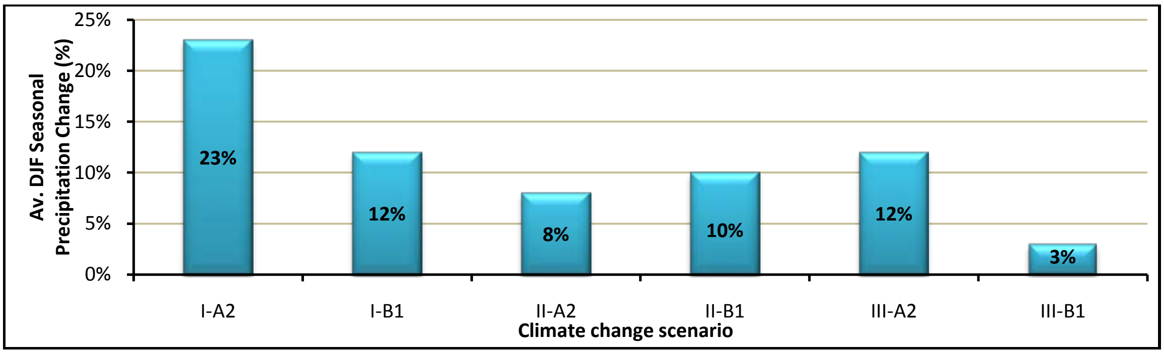

Figure 6.6: Average multi-model DJF (December, January and February) seasonal precipitation changes (%) (relative to 1950-1999 historical average).

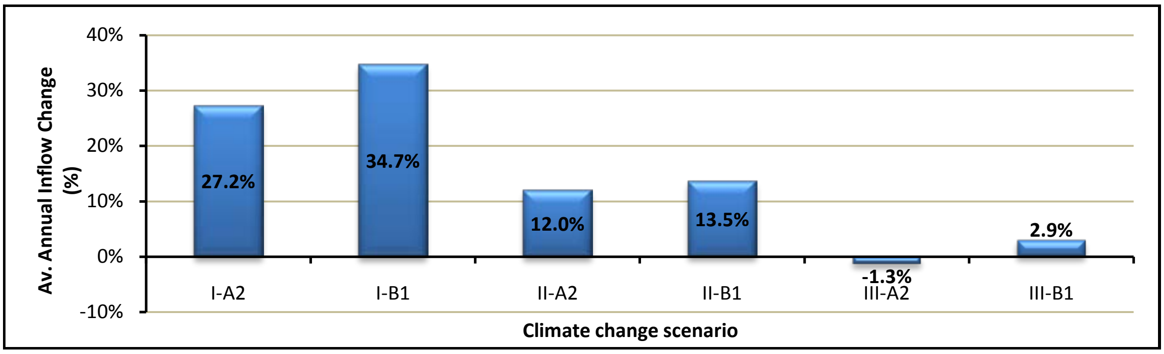

Figure 6.7: Average annual inflow changes (relative to the base case 2006).

6.5.2 Climate change impacts on the hydrodynamic characteristics

The global climate change effects on the hydrodynamic characteristics of Lake Nubia are presented in relation to the calibrated model results of the 2006 base case, which covers only two weeks in January 2006 (7.-19.) due to the limited availability of the historical data. Hence, the future climate change scenarios cover the predicted air temperature and inflow changes for the same period of the calibration process (January 7.–January 19.). Three hydrodynamic characteristics of the reservoir are investigated with respect to the climate change: water surface levels, evaporation water losses and thermal structure.

The predicted air temperature changes of the investigated area for different scenarios in the 21st century were used to modify the input meteorological file of the calibrated base case of 2006, which includes the measured records of air temperature of investigated area in January 2006.

Moreover, the predicted reservoir inflow changes for the same scenarios were used to modify the input inflow file of the calibrated base case of 2006, which includes the measured reservoir inflow in January 2006. The future initial conditions of water temperature, which are necessary for model simulation in the future, have been estimated by using the historical measured data base of Lake Nubia for different years. The presented results in this section were aggregately published by the author (Elshemy and Meon, 2010).

6.5.2.1 Results and discussion

Figure 6.8 shows the reservoir water levels change in %, relative to the stations water depths of the base case, for some selected scenarios. Five of six scenarios produce higher reservoir water levels than the base case, while the (III-A2) scenario, period III (2070-2099) with A2 global emission scenario, produces slightly lower reservoir water levels. The increase or decrease of the reservoir water levels varies from station to station depending on the station morphological characteristics. The figure shows that maximum water level change will be 3% for scenario I-B1, which has the maximum predicted annual inflow to the reservoir (+34.7%), at St. 4 which has the smallest cross section area, according to the reservoir bathymetry. These results are directly proportional to the hydrological model inflow predictions (as seen in Figure 6.7).

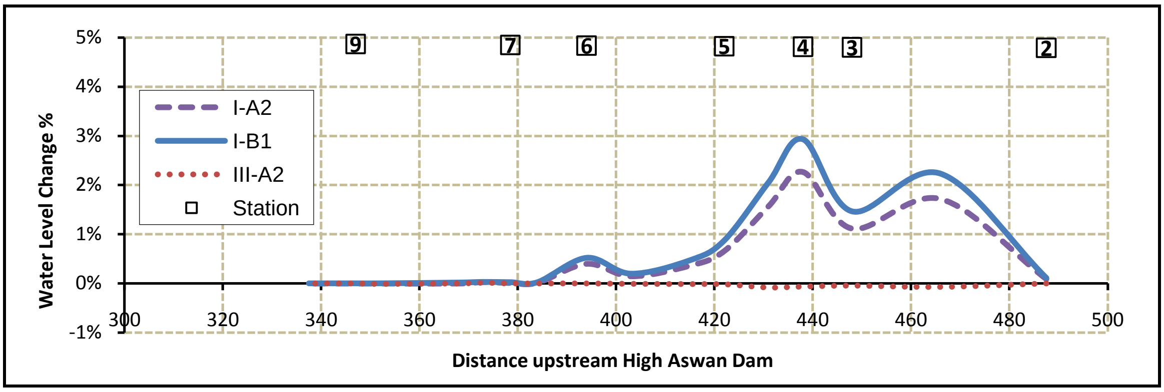

Figure 6.8: Lake Nubia water levels change (%) due to climate change for some selected scenarios.

The reservoir evaporation water losses change (%) is shown in Figure 6.9 for all scenarios. The figure shows that the maximum evaporation water losses change for the reservoir will be 7.7% for scenario III-A2 which has the maximum predicted air temperature change (3.9 °C). The evaporation water losses change is directly proportional to the air temperature change.

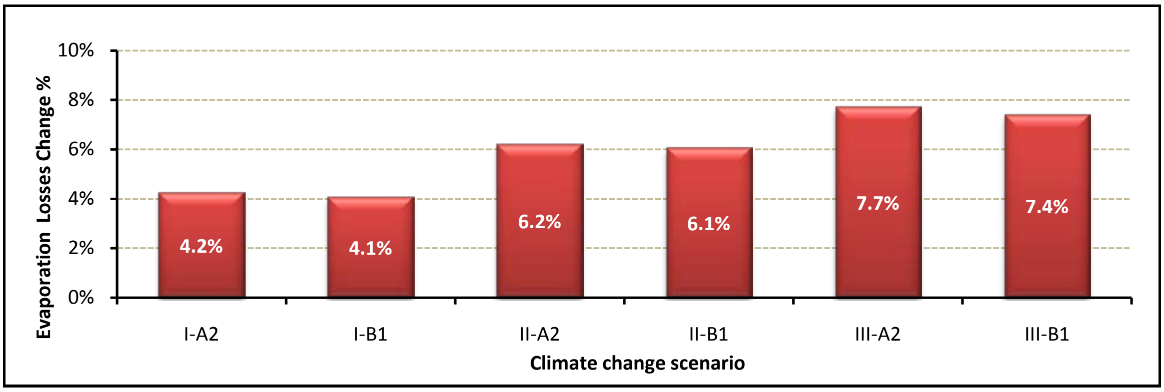

Figure 6.9: Lake Nubia evaporation water losses change (%) due to climate change for all scenarios.

Reservoir water temperature profiles for some selected scenarios and stations are presented in Figure 6.10 (Appendix 4 includes results at other different stations). ΔT is the average change (%) between the simulated base case and the simulated climate change scenario case. The reservoir water temperature is directly proportional to the air temperature and the reservoir surface area and inversely proportional to the predicted inflow (Chapra 1997). The reservoir water temperature change varies depending on the reservoir morphological and hydrological characteristics, for instance St. 5 has the smallest surface area and a higher water velocity (relative to the other stations) which enable it to have the minimum average difference (-7.42%) for scenario I-B1 which has the maximum predicted annual inflow (+34.7%). While the maximum average difference (5.91%) will be found at St. 7, which has a higher surface area, for III-A2 scenario which has the minimum predicted annual inflow to the lake (-1.3%).

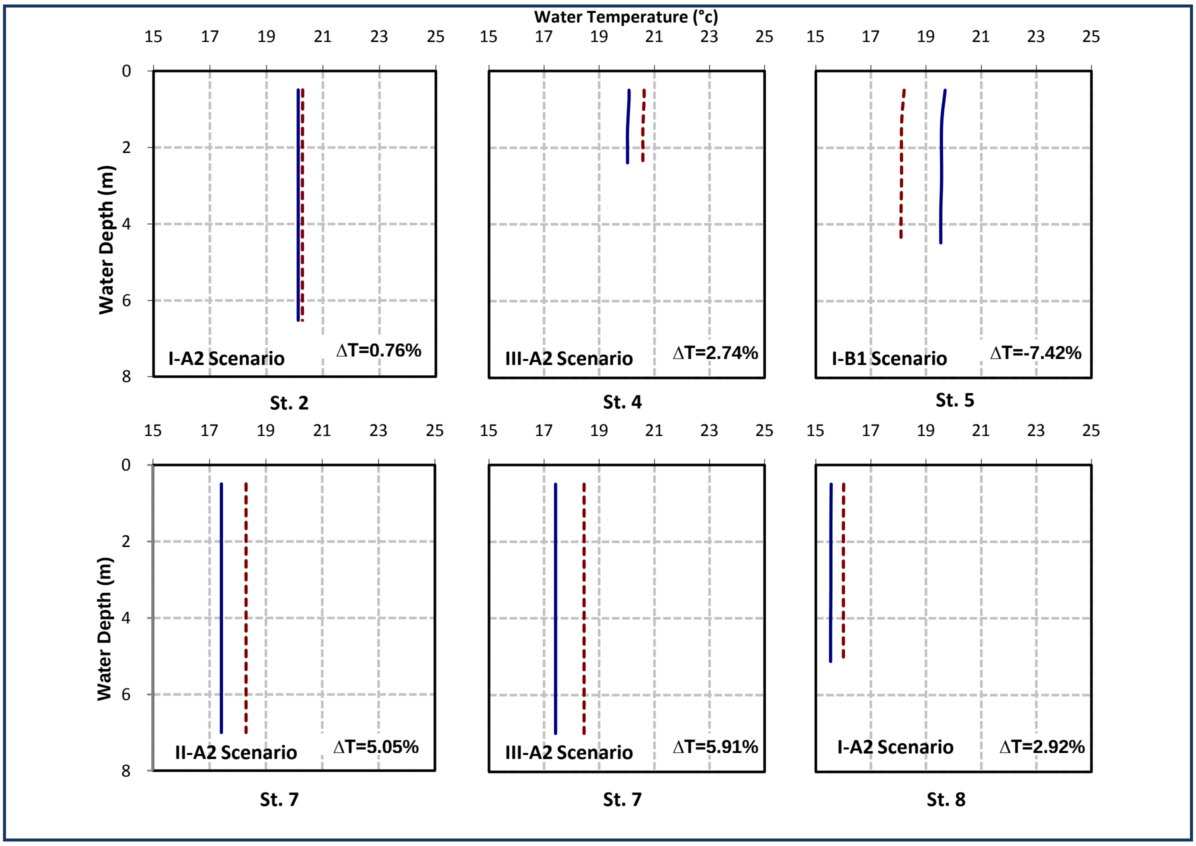

Figure 6.10: Lake Nubia water temperature profiles due to global climate change [- - -] for some selected scenarios and stations, compared with the simulated base case [—]

6.5.2.2 Sensitivity analysis

A sensitivity analysis has been done by using each of the predicted air temperature and inflow separately in the input files of the model to check its effect on water levels, evaporation water losses and reservoir water temperature.

Figure 6.11 presents the longitudinal water surface levels average difference profiles for some selected climate change scenarios. For all scenarios considering only the temperature effects, such as T I-A2 and T I-B1, there will be no change in the reservoir water levels. As can be seen in the current figure, the water level changes are directly proportional to the predicted inflow.

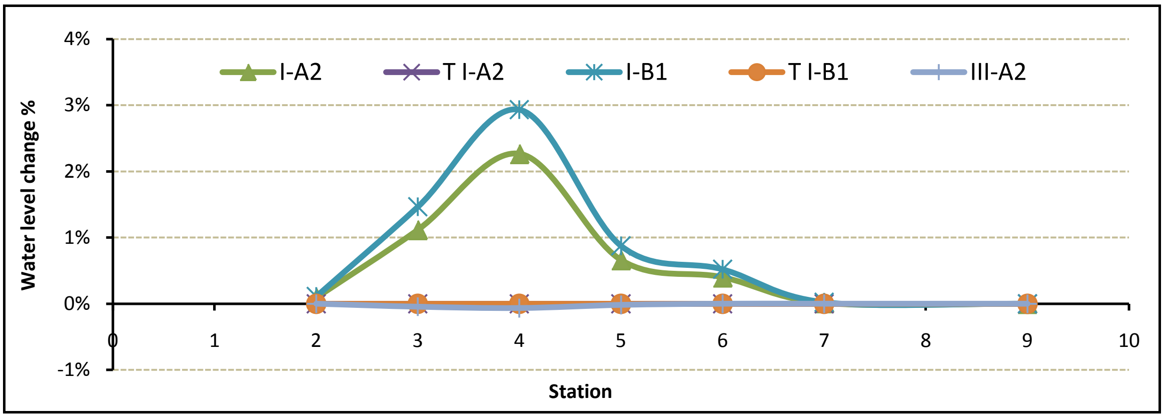

Figure 6.11: Lake Nubia water levels change (%) due to global climate change for some selected scenarios.

Lake Nubia evaporation water losses change (%) due to global climate change, air temperature effect only, for all scenarios can be seen in Figure 6.12. Comparison with Figure 6.9, which presents the evaporation water losses change due to the effect of both air temperature and inflow, shows higher values. This difference is caused by the direct relationship between evaporation water losses and the reservoir inflow which affect the morphological characteristics of the reservoir.

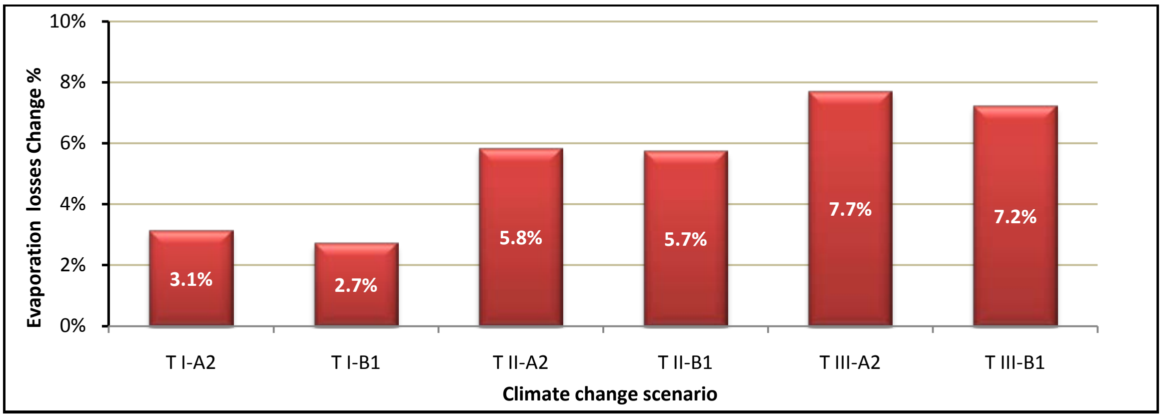

Figure 6.12: Lake Nubia evaporation water losses change (%) due to global climate change, air temperature effect only, for all scenarios.

Figure 6.13 presents longitudinal profiles of the average water temperature change (%) due to global climate change. Here, the air temperature effect was only considered. The average water temperature change longitudinal profiles due to the effect of both air temperature and reservoir inflow is shown in Figure 6.14.

It can be noticed that, as discussed before, the reservoir water temperature is directly proportional to the air temperature and inversely proportional to the predicted inflow. The reservoir water temperature change varies depending on the morphological characteristics which are influenced by the reservoir inflow.

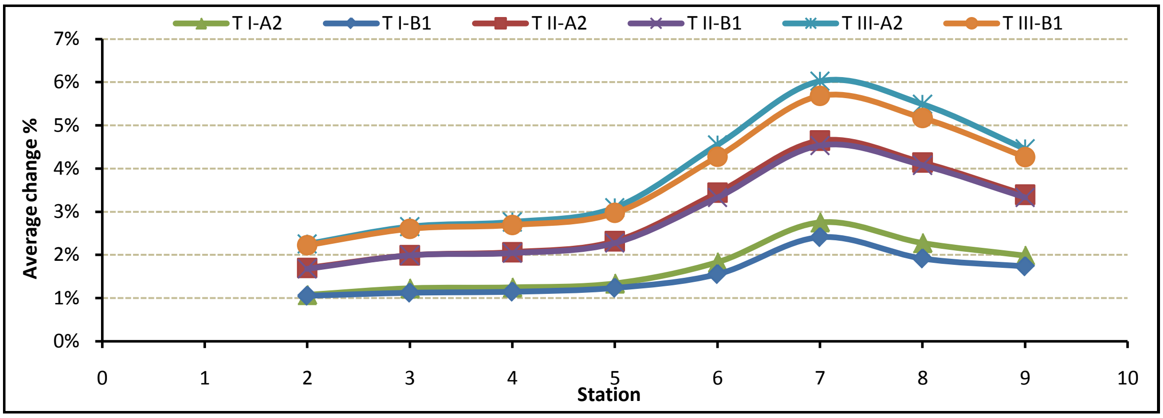

Figure 6.13: Lake Nubia average water temperature change (%) longitudinal profiles due to global climate change, air temperature effect only, for all scenarios.

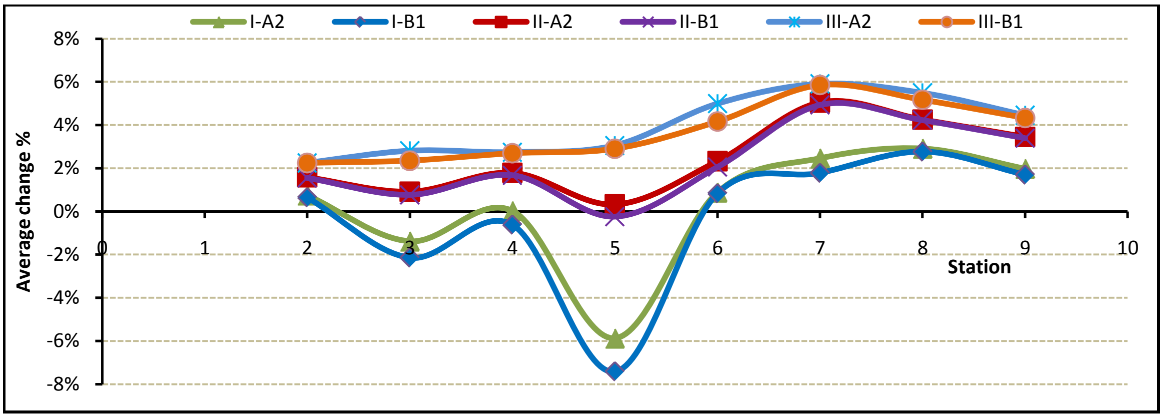

Figure 6.14: Lake Nubia average water temperature change (%) longitudinal profiles due to global climate change, air temperature and reservoir inflow, for all scenarios.

6.5.3 Climate change impacts on the water quality characteristics

The global climate change effects on the water quality characteristics of Lake Nubia are investigated relative to the calibrated model results of the 2006 base case, which covers only two weeks in January 2006 (7.-19.) due to the limited availability of the historical data. Hence, the future climate change scenarios cover the predicted air temperature and inflow changes for the same period of the calibration process (January 7.–January 19.). Eight water quality parameters of the reservoir were studied with respect to climate change: dissolved oxygen (DO), chlorophyll-a (Chl-a), ortho-phosphate (PO_4), nitrate-nitrite (NO_3-NO_2), ammonium (NH_4), total dissolved solids (TDS), total suspended solids (TSS) and potential of hydrogen (pH).

The input meteorological and inflow files of the calibrated base case of 2006 have been modified by using the predicted air temperature and reservoir inflow for different scenarios. The future initial conditions of water temperature, which are necessary for model simulation in the future, have been estimated by using the historical measured data base of Lake Nubia for different years. Dissolved oxygen (DO), the most important major water quality parameter, has been chosen to be researched due to the change of water temperature. A theoretical process algorithm has been simplified and further developed to modify the initial conditions input file of DO due to global climate change effects.

6.5.3.1 DO initial conditions estimation

Thomann and Mueller (1987) have reported the DO sources and sinks in the water body as follows:

The DO sources are:

1. Reaeration from the atmosphere.
2. Photosynthetic oxygen production.
3. DO in incoming tributaries or effluents.

While the internal sinks are:

4. Use of oxygen for respiration by aquatic plants.
5. Oxidation of carbonaceous waste materials.
6. Oxidation of nitrogenous waste materials.
7. Oxygen demand of sediments of water body.

Then the general mass balance equation for DO in a segment volume V can be written as:

$$\begin{aligned} V\frac{dDO}{dt} = {} & reaeration + (photosynthesis - respiration) \\ & - oxidation\ of\ CBOD, NBOD\ (from\ inputs) - sediment\ oxygen\ demand \\ & + oxygen\ inputs \\ & \pm oxygen\ transport\ (into\ and\ out\ of\ segment) \end{aligned} \tag{6.1}$$

The difference of DO concentration (ΔDO_T) due to the change of the water body water temperature from (T_{w1}) to (T_{w2}) can be calculated as follows:

$$\begin{aligned} \Delta DO_T = {} & [reaeration + (photosynthesis - respiration)]_{T_{w1}} \\ & - [reaeration + (photosynthesis \\ & - respiration)]_{T_{w2}} \end{aligned} \tag{6.2}$$

This simplified equation (Eq. 6.2) assumes that the other excluded sources and sinks of DO can be ignored; they slightly influenced by the water temperature changes from (T_{w1}) to (T_{w2}).

Reaeration DO calculation

The DO concentrations in water body due to reaeration (DO_a) can be calculated as follows (Chapra, 1997):

$$DO_a = DO_s(1 - e^{-k_a t}) \tag{6.3}$$

where:

DO_a DO concentration in water body due to atmospheric reaeration (mg/L).
DO_s DO saturated concentration (mg/L), which can be calculated as follows (Clescerl et al., 1999):

$$ln\, DO_S = -139.34411 + \frac{1.575701\times10^5}{T_W} - \frac{6.642308\times10^7}{{T_W}^2} + \frac{1.243800\times10^{10}}{{T_W}^3} - \frac{8.621949\times10^{11}}{{T_W}^4} \tag{6.4}$$

where:

T_w Water temperature (°K) ([°K]= [°C]+273.15)
t Time (d)

K_a Reaeration rate (d^{-1}), which can be calculated using O'Connor-Dobbins formula as follows (Chapra, 1997):

$$K_a = 3.93\ \frac{U^{0.5}}{H^{1.5}} \tag{6.5}$$

where:
U: Average stream velocity ($m\ sec^{-1}$)
H: Mean depth (m)

Photosynthesis and respiration DO calculations

The DO concentrations in water body due to photosynthesis (DO_p) and due to respiration (DO_r) can be calculated as follows (Thomann and Mueller, 1987):

$$DO_p = \left[a_{op} \times G_{max} \times (1.066)^{(T_w-20)} \times (chl-a)\right] \times G(I_a) \tag{6.6}$$

$$DO_r = a_{op} \times (0.10) \times (1.08)^{(T_w-20)} \times (chl-a) \tag{6.7}$$

Where:
a_{op} Ratio of (DO (mgl^{-1})/chl-a (μgl^{-1}), ranges (0.1 – 0.3)
G_{max} Maximum growth rate of the phytoplankton, ranges (1.5 – 3.0 /day)
T_w Water temperature (°C)
$chl-a$ Phytoplankton chlorophyll (μgl^{-1})
$G(I_a)$ Light attenuation factor over depth and one day (dimensionless), ranges (0 – 1.0)

The light attenuation factor can be calculated as follows:

$$G(I_a) = \frac{2.718f}{K_eH}\ [Exp(-\propto_1) - Exp(-\propto_\circ)] \tag{6.8}$$

For:

$$\propto_1 = \frac{I_a}{I_s} \times Exp(-K_eH) \quad , \qquad \propto_\circ = \frac{I_a}{I_s} \tag{6.9}$$

where:
f Photoperiod (sunlight fraction of day)
K_e Light extinction coefficient (m^{-1}), which can be calculated as follows (Chapra, 1997):

$$K_e = \frac{1.8}{SD}\ , \quad SD\ \ is\ Secchi\text{-}desk\ (m) \tag{6.10}$$

where:
H Layer water depth (m)
I_a Average solar radiation during the day ($ly\ d^{-1}$)

$$I_a = \frac{Mean\ daily\ solar\ radiation}{f} \tag{6.11}$$

I_s Light intensity at which plant growth is optimal, ranges (250 – 500 $ly\ d^{-1}$).

Then by using the previous simplified theoretical process algorithm, (DO_{nc}) initial conditions at a station No. n (St. n) due to change of water temperature of base case (T_{nb}) to (T_{nc}) can be calculated as follows:

$$DO_{nc} = DO_{nb} + \Delta DO_{(T_{nb}-T_{nc})} \tag{6.12}$$

where:

DO_{nc} Future DO concentration initial condition at St. n.
DO_{nb} Base case DO concentration initial condition at St. n.
$\Delta DO_{(T_{nb}-T_{nc})}$ DO concentration difference at St. n due to the change of water temperature at St. n from the base case (T_{nb}) to the future case (T_{nc}).

Equation (6.12) reduces the error of calculating the theoretical DO as much as possible.

DO algorithm calibration

To check the validity of the developed simplified theoretical process algorithm, a calibration process has been made using the measured data set of Lake Nubia of the year 2007 (Figure 6.15). The absolute mean error (AME) is 0.42 mg/L, which represents a reasonable error for initial condition estimation.

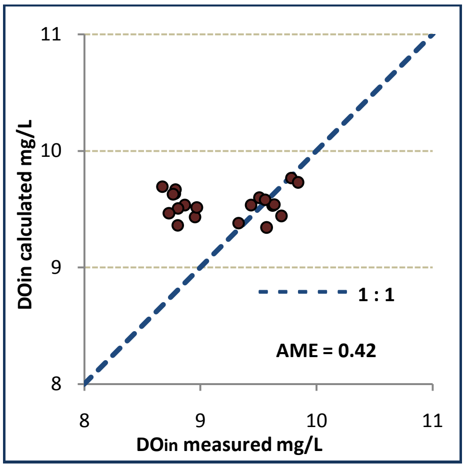

Figure 6.15: Calibration of initial DO calculation algorithm for Lake Nubia using a measured data set of the year 2007.

6.5.3.2 Results and discussion

Figures 6.16 – 6.18 show profiles of the Lake Nubia water quality characteristics due to global climate change for some selected scenarios and stations. ΔC is the average change (%) between the simulated base case of the year 2006 and the simulated scenario case. Appendix 4 includes results at other different stations.

DO change is inversely proportional to water temperature change and inflow change (refer to chapter 2. for literature). As can be seen in the current figures, the average changes of DO will decrease by 0.23%, 2.75% and 2.17% when water temperature average change increases by 1.68%, 1.78% and 5.49% at St. 4, St. 7 and St. 8 for scenarios II-B1, I-B1 and III-A2, respectively. While the inflow changes will be +13.5%, +34.7% and -1.3% for scenarios II-B1, I-B1 and III-A2, respectively.

The biological activity (i.e. photosynthesis and respiration) is affected by water temperature change and inflow change. As water temperature rises, the metabolic rates of the most water organisms increase. While, when inflow change increases, suspended solids increases affect the euphotic zone depth and then algal biomass. Moreover, inflow change affects the phtoplankton stability. Chlorophyll – a is an indirect measure of algal biomass, the negative changes of chlorophyll – a in the current figures (Figures 6.16 – 6.18) may return to the inflow change, such as at St. 7 for scenario I-B1 (inflow change will be 34.7%), or the decrease of DO, such as at St. 8 for scenario III-A2 where DO change will decrease by 2.17% which will affect the respiration and decomposition processes.

Inflow changes affect the nutrients concentrations in the water body. When inflow increases, surface erosion from the catchment (soil leaching) increases and then nitrate – nitrite, ammonium and phosphate concentrations will increase. Such as at St. 4 and St. 7 where inflow change increase by 13.5% and 34.7%, respectively. The ammonium and nitrate – nitrite concentration changes are controlled by the DO change which affect the nitrification process; when DO change decreases, nitrification process decreases and then ammonium concentration increases while nitrate – nitrite decreases. Increase in the nutrients concentrations can be noticed at St. 8 due to the decrease of algae biomass.

Suspended solids change is directly proportional mainly to the inflow change, as can be seen in the presented figures (Figures 6.16 – 6.18). When inflow increases, inflow velocity increases and settlement velocity of suspended solids decreases, hence suspended solids concentration increases.

Potential of hydrogen (pH) is affected by water temperature, dissolved solids, alkalinity and total inorganic carbon. At St. 4 and St. 7 for scenarios II-B1 and I-B1, respectively, inflow changes increase which increase total solids concentrations and different ions (by soil leaching), hence resulted in decreasing of pH. For scenario III-A2 at St. 8, pH will decrease due to increase of total inorganic carbon average change by 30.44% as a result of water temperature increase which increase the metabolic activity.

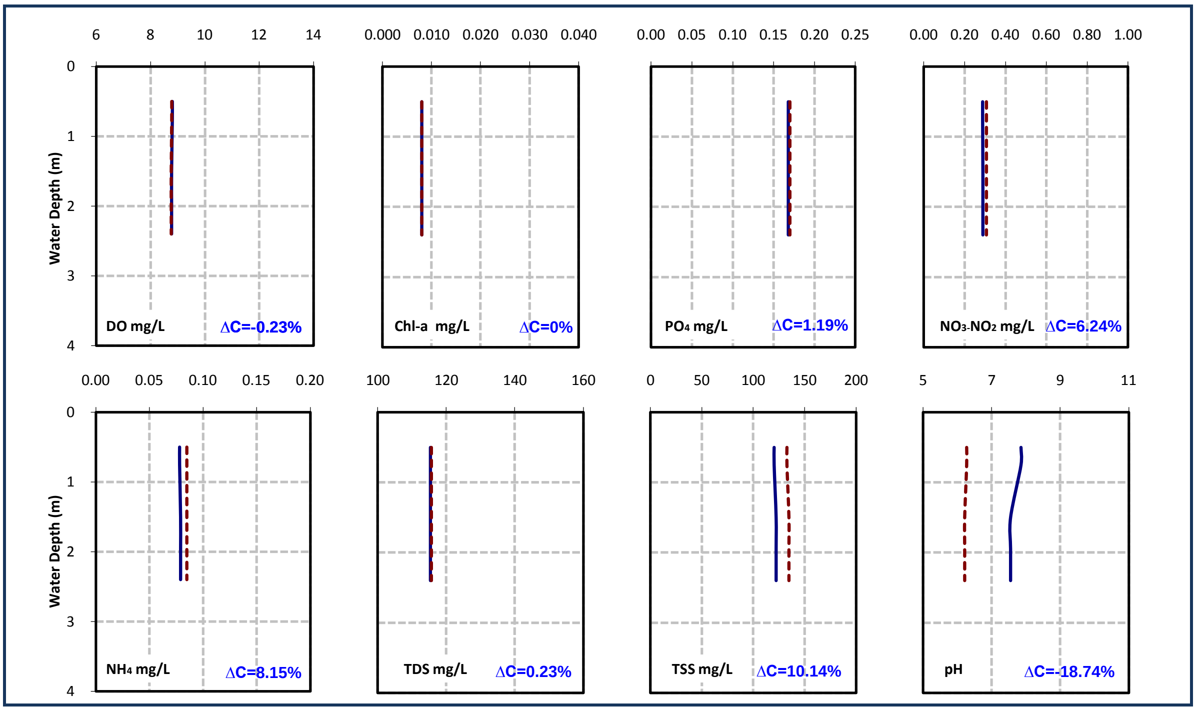

Figure 6.16: Lake Nubia water quality characteristics profiles due to global climate change [- - -] for II-B1 scenario at St. 4, compared with the simulated base case [—].

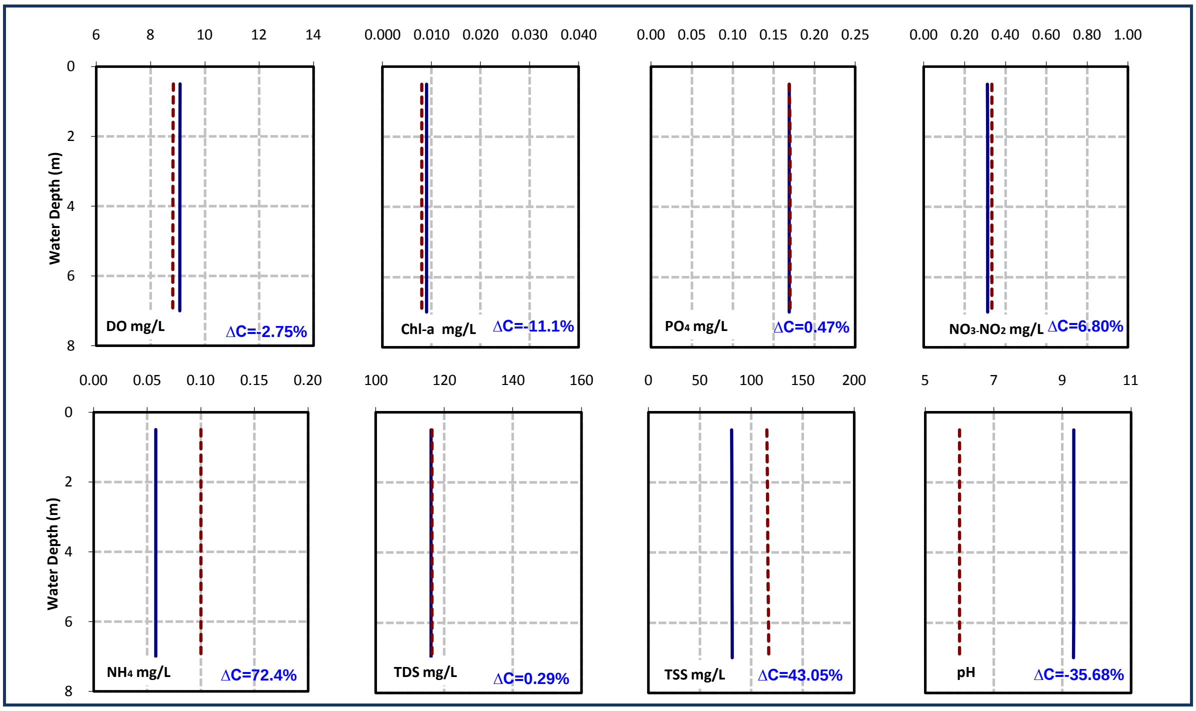

Figure 6.17: Lake Nubia water quality characteristics profiles due to global climate change [- - -] for I-B1 scenario at St. 7, compared with the simulated base case [—].

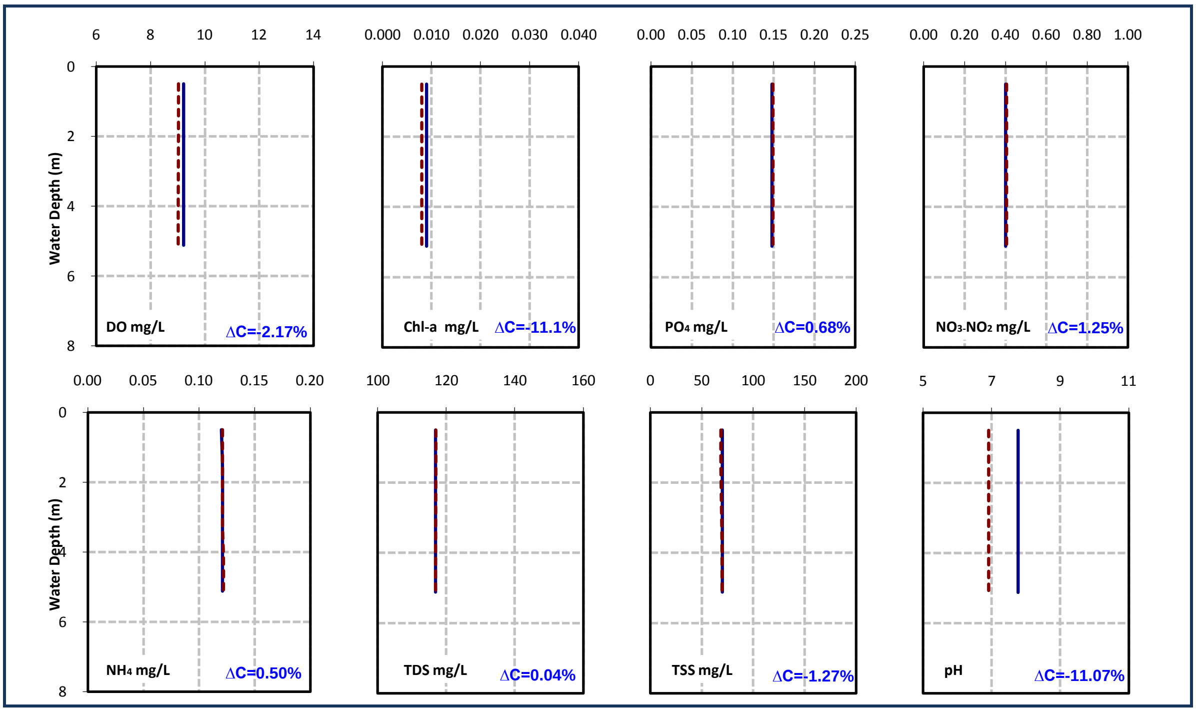

Figure 6.18: Lake Nubia water quality characteristics profiles due to global climate change [- - -] for III-A2 scenario at St. 8, compared with the simulated base case [—].

6.5.3.3 Sensitivity analysis

A sensitivity analysis has been done by using each of the predicted air temperature and inflow separately in the input files of the model to check its effect on water quality characteristics of Lake Nubia. Figures 6.19 and 6.20 present Lake Nubia longitudinal DO average difference profiles for different climate change scenarios for change of air temperature only, and change of both air temperature and inflow, respectively. As discussed before, the DO change is inversely proportional to water temperature and inflow changes. DO changes longitudinal profiles (Figures 6.19 and 6.20) are in accordance with water temperature changes longitudinal profiles (Figures 6.13 and 6.14), where DO concentration decreases as water temperature increases. When inflow increases, DO concentration decreases. The maximum DO average decrease, for air temperature change only (Figure 6.19), will be 2.3%, relative to the base case of the year 2006, at St. 8 for III-A2 scenario which has the maximum predicted air temperature change (3.9 °C). While for change of air temperature and inflow changes (Figure 6.20), the maximum DO average decrease will be 3.9% at St. 8 for I-B1 scenario which has the maximum predicted inflow change (34.7%). The maximum DO average increase will be 2.7% and 2.9% for I-B1 scenario which has the minimum predicted air temperature change (1.0 °C), for change of air temperature only (Figure 6.19) and both air temperature and inflow (Figure 6.20), respectively, at St. 9 which has the maximum surface area and water depth of Lake Nubia; inflow change has the lowest effect on water velocity at this station.

Lake Nubia water quality characteristics profiles due to global climate change, change of air temperature only, for II-B1 scenario at St. 4 are presented in Figure 6.21. The severe effect of inflow change on most of the water quality characteristics of the reservoir can be observed when Figure 6.21 is compared to Figure 6.16 which exhibits the both changes of air temperature and inflow effects on the water quality characteristics of the reservoirs.

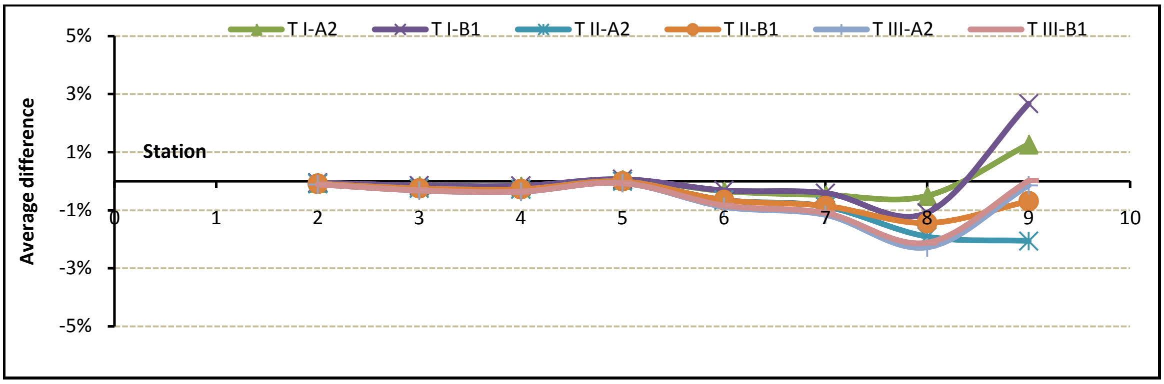

Figure 6.19: Lake Nubia longitudinal DO average change (%) profiles for different climate change scenarios for change of air temperature only.

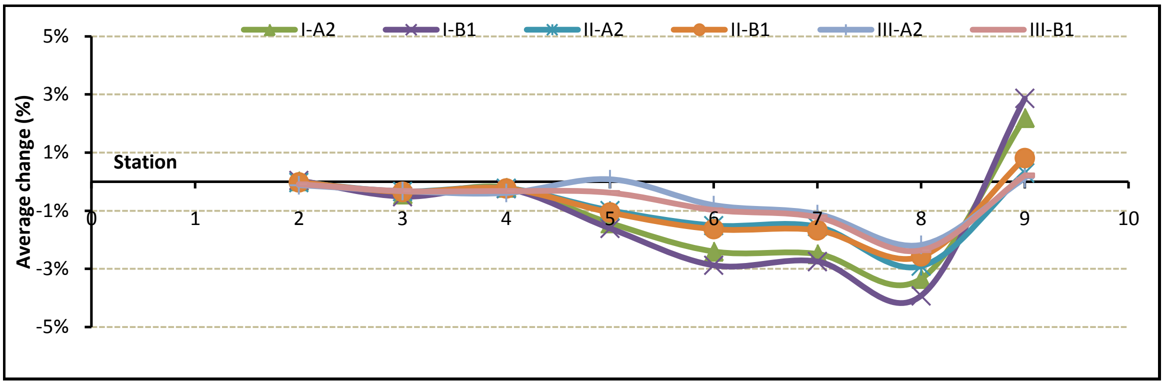

Figure 6.20: Lake Nubia longitudinal DO average change (%) profiles for different climate change scenarios for change of both air temperature and inflow.

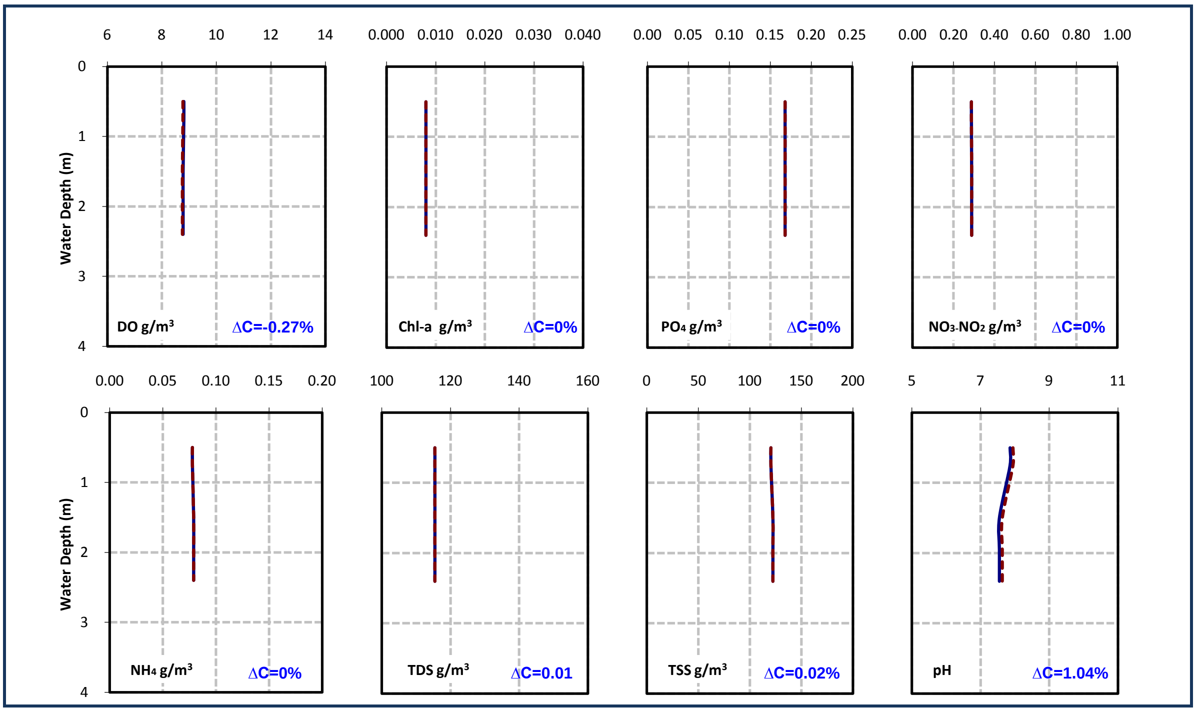

Figure 6.21: Lake Nubia water quality characteristics profiles due to global climate change [- - -], change of air temperature only, for II-B1 scenario at St. 4, compared with the simulated base case [—].

6.5.4 Climate change impacts on Lake Nubia water quality indices

Two water quality indices (WQIs), NSF WQI and CCME WQI, have been implemented to assess the water quality state of Lake Nubia, refer to the former chapter (chapter 5). The effect of global climate change on both water quality indices NSF and CCME has been investigated and presented relative to the calibrated model results of the 2006 base case, which covers only two weeks in January 2006 (7.-19.) due to the limited availability of the historical data. Hence, the future climate change scenarios cover the predicted air temperature and inflow changes for the same period of the calibration process (January 7.–January 19.). The presented results in this section were aggregately published by the author (Elshemy and Meon, 2011).

6.5.4.1 NSF WQI

Eight parameters have been used to apply the NSF WQI to Lake Nubia. The implemented parameters are: dissolved oxygen, fecal coliform, pH, temperature change, total phosphate, nitrate, turbidity and total solids. Figure 6.22 shows the effect of climate change on the average NSF WQI of Lake Nubia. The change of average NSF WQI due to climate change is controlled by the change of inflow more than the change of air temperature; where the inflow change affect most of the water quality used parameters. The climate change will slightly decrease the average NSF WQI of Lake Nubia; the maximum decrease will be 3%, relative to the base case, for scenario I-B1 which has the maximum predicted inflow change (+34.7%).

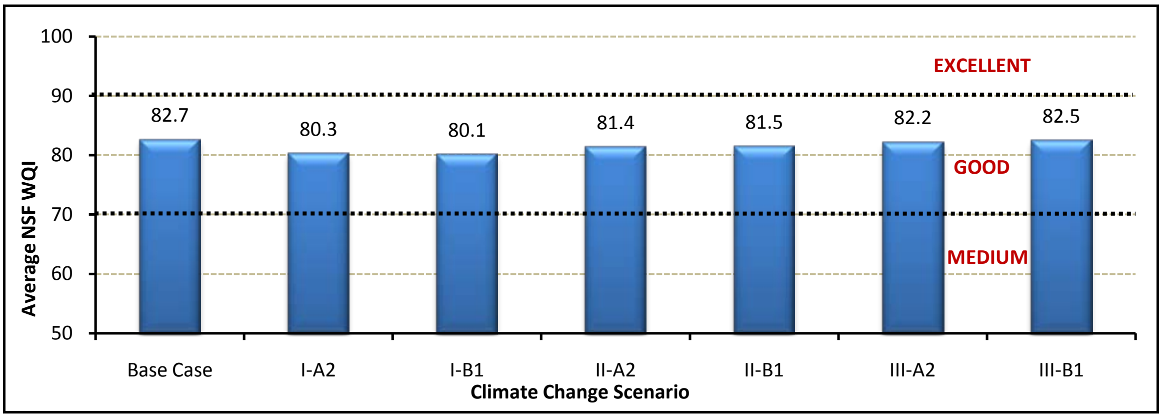

Figure 6.22: Lake Nubia average NSF WQI for different climate change scenarios.

Figure 6.23 presents the longitudinal profiles of Lake Nubia for I-B1 and III-A2 climate change scenarios which reflect the maximum and minimum predicted flow scenarios, respectively. For I-B1 scenario, the maximum NSF WQI change will be -6.2% at St. 4 because of the increase of suspended solids and fecal coliform concentrations and the decrease of pH due to the increase of inflow by 34.7%. While, for the low predicted inflow III-A2, the maximum NSF WQI change will be -2.4% at St. 6 because of the water temperature and pH increase and DO decrease. As explained before, the climate change affects the reservoir water quality characteristics which in turn influence water quality state of the reservoir.

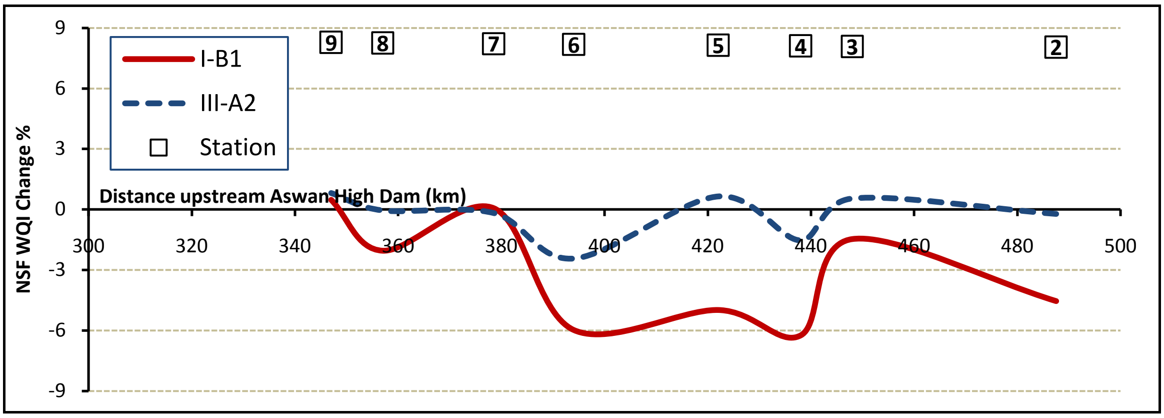

Figure 6.23: Lake Nubia average NSF WQI longitudinal profiles for some selected climate change scenarios.

6.5.4.2 CCME WQI

CCME WQI was developed to estimate the overall water quality status of Lake Nubia. Seven parameters were used to apply the CCME WQI to Lake Nubia. The used parameters are: Dissolved oxygen, total dissolved solids, nitrate-nitrite, ammonium, total phosphorus, fecal coliform and pH. Figure 6.24 presents the effect of climate change on the CCME WQI of Lake Nubia for different climate change scenarios. Predicted inflow change affects most of water quality parameters, which in turn influence the reservoir water quality state, more than predicted air temperature change. For CCME WQI, climate change will slightly decrease CCME WQI of Lake Nubia; the maximum decrease will be 2.7% at I-B1 scenario, where the predicted inflow will be the maximum change (+34.7%). While the minimum decrease will be 0.3% at III-A2 scenario, where the predicted inflow change will be the minimum (-1.3%) and the predicted air temperature change will be the maximum (3.9 °C).

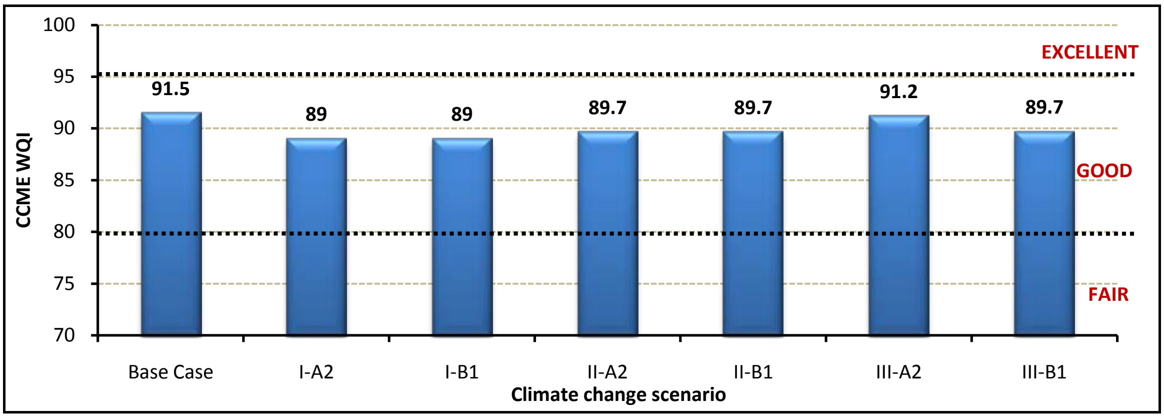

Figure 6.24: Lake Nubia CCME WQI for different climate change scenarios.

6.5.5 Climate change impacts on Lake Nubia trophic status Indices

Carlson Trophic Status Index (TSI) and LAWA Total Index (TI) have been applied to assess the trophic status of Lake Nubia, refer to the previous chapter (chapter 5). The effect of global climate change on trophic status indices Carlson TSI and LAWA TI has been investigated and presented relative to the calibrated model results of the 2006 base case, which covers only two weeks in January 2006 (7.-19.) due to the limited availability of the historical data. Hence, the future climate change scenarios cover the predicted air temperature and inflow changes for the same period of the calibration process (January 7.–January 19.).

6.5.5.1 Carlson TSI

Carlson TSI was used for evaluating the trophic state of Lake Nubia. Two water quality parameters were used to estimate Carlson TSI; total phosphorus (TP) and chlorophyll – a (Chl-a). The effect of climate change on average Carlson TSI of Lake Nubia for different climate change scenarios can be seen in Figure 6.25. Climate change will slightly affect average Carlson TSI, the maximum change will be a decrease by 0.5% for II-A2 scenario due to the slight change of TP and Chl-a.

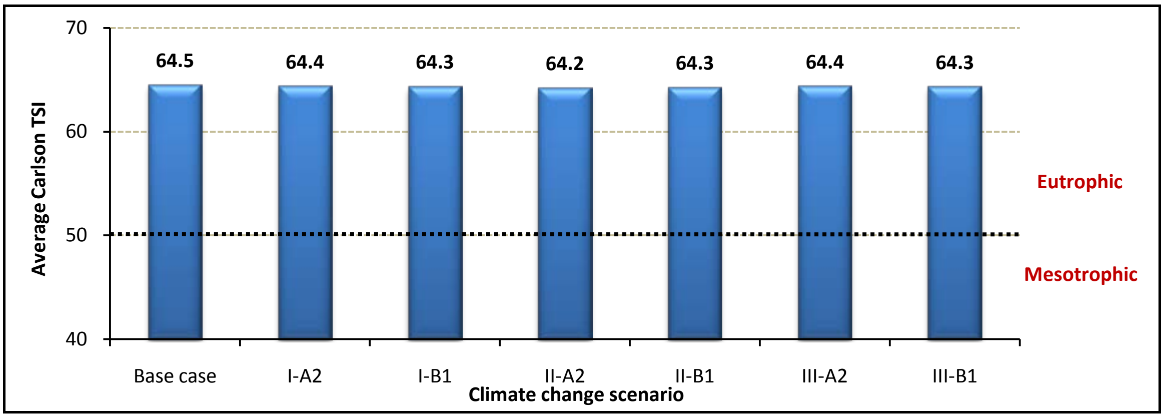

Figure 6.25: Lake Nubia average Carlson TSI for different climate change scenarios.

The longitudinal profiles of average Carlson TSI of Lake Nubia is presented in Figure 6.26 for I-B1 and III-A2 climate change scenarios which exhibit the maximum and minimum predicted flow scenarios, respectively. As shown in the figure, average Carlson TSI change is controlled by the climate change effect on Chl-a concentration more than TP concentration at different stations.

For I-B1 scenario, TP concentrations will slightly increase; maximum change will be 1.17% at St. 4, while Chl-a concentrations will decrease; the maximum decrease will be 12.5% at St. 3. For St. 7 and St. 8, Chl-a concentrations will decrease by 11% while TP concentrations will slightly increase by 0.6%. For III-A2 scenario, chl-a concentrations will decrease by 11% at St. 8 and St. 9, while TP concentrations will increase by 0.8% at St. 9.

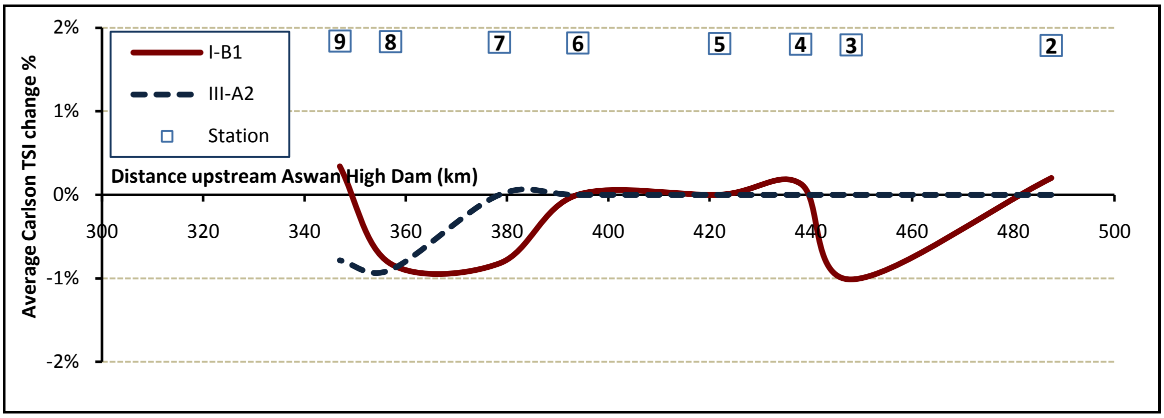

Figure 6.26: Lake Nubia average Carlson TSI longitudinal profiles for some selected climate change scenarios.

6.5.5.2 LAWA TI

As in Carlson TSI, two water quality parameters were used to estimate LAWA TI for Lake Nubia; total phosphorus (TP) and chlorophyll – a (Chl-a). Figures 6.27 and 6.28 present Lake Nubia LAWA TI for different climate change scenarios and longitudinal profiles for some selected scenarios, respectively. The results show the same slight effect of the climate change at the same stations as Figures 6.25 and 6.26 of Carlson TSI, this is because both indices are relied on the same parameters, TP and Chl-a. As can be indicated in Figure 6.27, the maximum change will be a decrease by 1.4% for II-A2 scenario. While in Figure 6.28, the maximum decreases will be 3% at St. 3 and 2.5% at St. 8, for I-B1 and III-A2 scenarios, respectively.

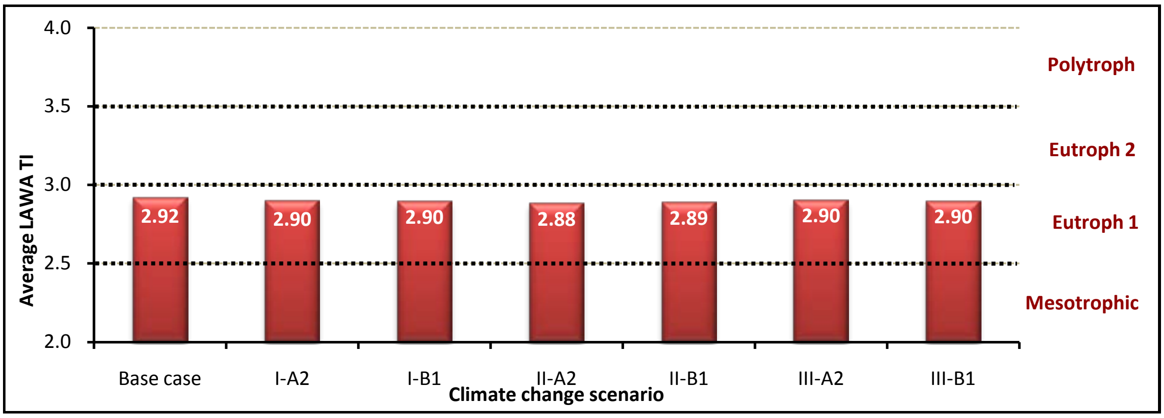

Figure 6.27: Lake Nubia LAWA TI for different climate change scenarios.

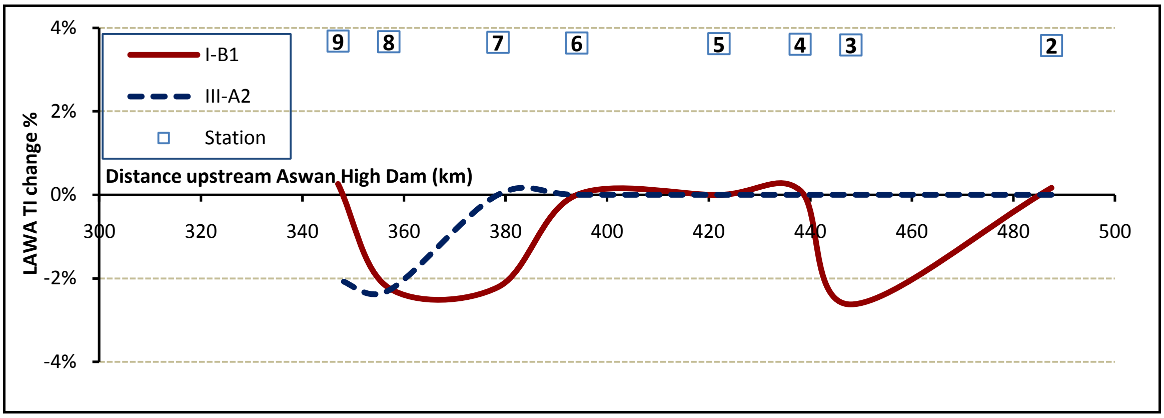

Figure 6.28: Lake Nubia LAWA TI longitudinal profiles for some selected climate change scenarios.

6.5.6 Conclusions

One of the main goals of this research work is to roughly quantify the potential impact of the global climate change on hydrodynamic and water quality characteristics of Lake Nubia. In doing so, we recognize the limitations of this modeling approach including the uncertainty in GCMs output, the hydrological model and the hydrodynamic and water quality model. Thus, the results presented here are an indication of the effects of climate change on hydrodynamic and water quality characteristics of Lake Nubia, and are not intended to be exact predictions.

The impacts of climate change on hydrodynamic and water quality characteristics of Lake Nubia have been investigated using a proposed hydrodynamic and water quality model, which simulates three periods: I [2010-2039], II [2040-2069] and III [2070-2099] - with two emission scenarios, A2 and B1, for each period, including the average of eleven GCMs outputs. A theoretical process algorithm has been simplified and further developed to modify the initial conditions input file of dissolved oxygen due to global climate change effects. A sensitivity analysis has been done by using each of predicted air temperature and inflow separately in the proposed model input files to check its effect on the hydrodynamic and water quality characteristics. Moreover, the climate change effects on water quality and trophic status indices have been studied. The global climate change effects are presented relative to the calibrated model results of the year 2006 base case.

It is emphasized that the calibration and the scenarios cover only a short period of two weeks of simulation. This is due to the limited availability of historic data (chapter 3). Consequently the results and quantified effects do not reflect the variability of hydrodynamics and water quality extending over one year or even longer periods. Such an extension is recommended as essential future work.

The effects of global climate change on three hydrodynamic characteristics of Lake Nubia have been investigated. The studied hydrodynamic characteristics are: water levels, evaporation water losses and reservoir thermal structure. The results of the climate change show that, there are significant impacts on the examined characteristics. Thus, it can be concluded that:

1. Water level changes are directly proportional to the hydrological model inflow predictions. The maximum water level change will be 3% for scenario I-B1 at St. 4.

2. The evaporation water losses change is directly proportional to the air temperature change. The evaporation water losses are higher than the base case for all scenarios, the maximum evaporation water losses change for the reservoir will be 7.7% for scenario III-A2.
3. The reservoir water temperature changes are directly proportional to the air temperature changes and the reservoir surface area and inversely proportional to the predicted inflow changes. The maximum water temperature change will be 5.9% for scenario III-A2 at St. 7, while the minimum change will be -7.4% for scenario I-B1 at St. 5.

The global climate change effects on the water quality characteristics of Lake Nubia are investigated relative to the calibrated model results of the 2006 base case. Eight water quality parameters of the reservoir were studied with respect to climate change: dissolved oxygen (DO), chlorophyll-a (Chl-a), ortho-phosphate (PO_4), nitrate-nitrite (NO_3-NO_2), ammonium (NH_4), total dissolved solids (TDS), total suspended solids (TSS) and potential of hydrogen (pH). From the results, it can be concluded that climate change has a clear impacts on the investigated parameters as follows:

1. The change of DO is inversely proportional to water temperature and inflow changes. The maximum DO average decrease will be 3.9% at St. 8 for I-B1, while the maximum DO average increase will be 2.9% for I-B1 scenario at St. 9.
2. Biological activity (i.e. photosynthesis and respiration) is affected by water temperature change and inflow change. Chl-a will decrease by 11% at St. 7 and St. 8 for scenarios I-B1 and III-A2, respectively.
3. Inflow changes affect the nutrients concentrations in the water body. For the maximum predicted inflow scenario I-B1 at St. 7, PO_4 will have an increase of 0.5% while NH_4 and NO_3-NO_2 will increase by 72.4% and 6.8%, respectively.
4. Suspended solids change is directly proportional mainly to the inflow change. For the maximum predicted inflow scenario I-B1 at St. 7, TSS and TDS will increase by 43% and 0.3%, respectively.
5. Potential of hydrogen (pH) is affected by water temperature, dissolved solids, alkalinity and total inorganic carbon. When inflow changes increase, total solids concentrations with different ions (by soil leaching) increase, hence pH decreases such as for I-B1 scenario at St. 7, pH will decrease by 35.7%.
6. Sensitivity analysis results show the severe effect of inflow change on water temperature and most of the water quality characteristics of the reservoir. For water temperature, the sensitivity analysis results show that there will be no decreases at all when the future estimated inflow changes are ignored, while the maximum DO average decrease will be 2.3% for the same conditions.

The effect of global climate change on both water quality indices NSF and CCME has been investigated. From the results it can be concluded that:

1. The change of both NSF WQI and CCME WQI due to climate change is controlled by changing of inflow more than the change of air temperature as inflow change affect most of the water quality used parameters, which in turn affect water quality state of the reservoir.

2. The climate change will slightly decrease the average NSF WQI of Lake Nubia. For scenario I-B1, the maximum decrease of average NSF WQI will be 3%, while the maximum NSF WQI change will be -6.2% at St. 4.
3. For CCME WQI, climate change will slightly decrease CCME WQI of Lake Nubia; the maximum decrease will be 2.7% at I-B1 scenario.

The effect of global climate change on both trophic status indices Carlson TSI and LAWA TI has been investigated. Based on the results following conclusions can be considered:

1. Carlson TSI and LAWA TI changes are controlled by the climate change effect on Chl-a concentrations more than total phosphorus (TP) concentrations at different stations. As both indices are based on the same parameters, they will have, approximately, the same slight climate change effects.
2. Climate change will slightly affect Carlson TSI, the maximum change of average Carlson TSI will decrease by 0.5% for II-A2 scenario. While the maximum decrease of Carlson TSI will be 1% at St. 3 for I-B1 scenario.
3. For LAWA TI, the maximum change of average LAWA TI will decrease by 0.7% for II-A2 scenario. While the maximum decrease of LAWA TI will be 3% at St. 3 for I-B1 scenario.

6.5.7 Recommendations for future work

1. Past long-term records of hydrodynamic and water quality characteristics of the Aswan High Dam (AHD) reservoir are urgently required to investigate in more detail the climate change impacts based on a much longer period of simulation than it could be realized in this research work due to the very limited historical data base being presently available.
2. A regional climate model for the Nile basin should be developed to estimate changes of different meteorological parameters. The results of this regional climate model may be used to simulate the runoff pattern by using a suitable hydrological model.
3. Daily records of hydrodynamic and water quality characteristics of the AHD reservoir for one year, at least, should be examined to investigate climate change impacts of the reservoir thermal structure and the water column stability.

7. Extended summary, conclusions and recommendations

7.1 Extended summary and conclusions

The main objectives of the current research work were to develop a hydrodynamic and water quality model for a large reservoir which represents the characteristics of a semi-arid region. For this, the Lake Nubia being the southern part of Aswan High Dam reservoir was selected. Then the adapted model was used to evaluate the water quality and trophic status of the reservoir for different scenarios (in the present and future) by developing different water quality and trophic status indices. Moreover, the proposed model was used to investigate the impacts of global climate change on different hydrodynamic and water quality characteristics for different scenarios in the future.

The present hydrodynamic and water quality characteristics of Lake Nubia were investigated by applying the laterally averaged, two-dimensional hydrodynamic and water quality modeling system (code) CE-QUAL-W2. The model was calibrated and verified with data from January 2006 and February 2007, respectively. The investigated hydrodynamic characteristics were the water surface level and the water temperature. Water quality characteristics were dissolved oxygen (DO), chlorophyll-a, ortho-phosphate, nitrate-nitrite, ammonium, total dissolved solids, total suspended solids and potential of hydrogen (pH). The hydrodynamic model simulation results show a good agreement with the observed water surface levels and the measured water temperature profiles at various locations and dates. For water surface levels, the AME (absolute mean error) was 0.088 m for the calibration period. For the thermal structure, the average AME was 0.26 °C and 0.37 °C for the calibration and verification periods, respectively. Spatial and temporal results of the water quality model closely match the measured vertical profiles of the water quality constituents. For dissolved oxygen, for example, the average AME was 0.275 mg/L and 0.491 mg/L for the calibration and verification periods, respectively. The vertical profiles (measured and simulated) of water temperature and dissolved oxygen closely match the typical thermal distribution for warm monomictic lakes in winter.

Two water quality indices, NSF WQI and CCME WQI, and two trophic status indices, Carlson TSI and LAWA TI, were calculated to assess the water quality and trophic status of the investigated case study, Lake Nubia, during low flood period, January 2006. The CCME WQI was developed according to the Egyptian water quality standards for surface waterways. Moreover, the proposed hydrodynamic and water quality model of Lake Nubia was used to estimate the four indices during the same period, January 2006, to verify the model validity. According to the developed water quality indices results, measured and simulated, Lake Nubia has a good water quality state during the low flood period. The modified CCME WQI, based on measured data, indicates that the Lake Nubia water quality state is excellent, according to the Egyptian water quality standards for surface waterways. Results of applied trophic status indices, Carlson TSI and LAWA TI, show that the Lake Nubia trophic status is eutrophic during low flood period. The proposed hydrodynamic and water quality model shows a good agreement between the indices estimation based on measured and simulated parameters.

One of the main goals of this research work was to roughly quantify the potential influence that global climate change may have on hydrodynamic and water quality characteristics of Lake Nubia. These impacts have been investigated using the proposed hydrodynamic and water quality model of Lake Nubia, for three periods: I [2010-2039], II [2040-2069] and III [2070-2099] - with two emission scenarios, A2 and B1, for each period, including the average of eleven global climate models outputs. To estimate the future initial conditions of the proposed model, a theoretical process algorithm was simplified, developed and calibrated. In addition, a sensitivity analysis for the future climate estimates was conducted. Moreover, the effects of future climate change on water quality and trophic status of Lake Nubia were studied. The results of this study are presented relative to the calibrated model results of the year 2006 as a base case.

The investigated hydrodynamic characteristics were: water surface levels, evaporation water losses and reservoir thermal structure. The results show that there will be significant impacts of the climate change on the examined hydrodynamic characteristics. Water level changes are directly proportional to the future estimated inflow changes; the maximum water level change will be 3%. As evaporation water losses change is directly proportional to the air temperature change, evaporation water losses will be higher than the base case for all scenarios; the maximum evaporation water losses change for the reservoir will be 7.7%. For thermal structure, the reservoir water temperature changes are directly proportional to the air temperature changes and the reservoir surface area and inversely proportional to the predicted inflow changes. The maximum water temperature increase will be 5.9%, while the maximum decrease will be -7.4%.

Eight water quality parameters of the reservoir were studied with respect to climate change: dissolved oxygen, chlorophyll-a, ortho-phosphate, nitrate-nitrite, ammonium, total dissolved solids, total suspended solids and pH.

From the results, it can be concluded that the climate change will have a clear impact on the investigated parameters. Dissolved oxygen change is inversely proportional to water temperature and inflow changes. The maximum DO average decrease will be 3.9%, while the maximum DO average increase will be 2.9%. For nutrients, ortho-phosphate concentration, the increase will be 0.5% while ammonium and nitrate-nitrite will increase by 72.4% and 6.8%, respectively for the maximum predicted inflow scenario at St. 7 of the case study.

Sensitivity analysis results show the severe effect of inflow change on water temperature and most of the water quality characteristics of the reservoir. For water temperature, the results show that there will be no decreases at all, when the future estimated inflow changes are ignored. The maximum DO average decrease will be 2.3% for the same conditions.

The effect of global climate change on two water quality indices, NSF and CCME, and two trophic status indices, Carlson TSI and LAWA TI, were investigated. The results show that future climate change will slightly affect the investigated indices. The maximum NSF WQI change will be -6.2%, while the maximum decrease will be 2.7% for CCME WQI. For Carlson TSI, the maximum decrease will be 1%, while it will be 3% for LAWA TI.

7.2 Recommendations

Based on the study results, the recommendations for future research work are as follows:

1. The –by far- most demanding challenge which has been faced during this research work was the data limitation. For future research, an effective continuously water quality monitoring program, which provides daily or weekly water quality records – at least, for a longer period of 1 to 2 years –, is essential.
2. Although CE-QUAL-W2 simulates well the hydrodynamic and water quality characteristics of the reservoir, a comparative study is recommended using a three-dimensional code, if more data will be available in the future.
3. A long time trend water quality and trophic status study for the AHD reservoir should be conducted using a detailed database. Such a study will control the water quality management of the reservoir.
4. More accurate water quality and trophic status indices should be developed in particular for sub-tropical lakes and reservoirs such as the AHD reservoir.
5. The investigated reservoir zones should be assigned to different water uses according to its water quality and trophic states.
6. Past long-term records of hydrodynamic and water quality characteristics of the Aswan High Dam reservoir should be examined to investigate the impacts of the climate change.
7. Daily records of hydrodynamic and water quality characteristics of the AHD reservoir for at least one year should be examined to investigate the impacts of the climate change on the reservoir thermal structure and water column stability.
8. A regional climate model for the Nile basin should be developed to estimate changes of different meteorological parameters. The results of this regional climate model may be used to simulate the runoff pattern by using a suitable hydrological model.

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Appendix 1: Sample data

A1.1 Measured data during calibration period (January 2006)

Table A1.1: Field information of Lake Nubia at some selected stations during January 2006.

Site name	Station number	Distance upstream AHD (km)	Date of sampling	Time of sampling	Total depth (m)	Air temperature (°C)
ElDaka (23)	St. 1	487.5	7th	15.00	5.50	30.50
Kagnarty (6)	St. 6	394.0	13th	7.15	2.50	14.00
Dabarosa (22)	St. 10	337.5	19th	9.00	30.35	16.00

Table A1.2: Field measurments of different chemical and physical parameters of Lake Nubia at some selected stations during January 2006.

Station	Position	pH	DO (mg/L)	Water Temperature (°C)	Turbidity (NTU)	Secchi depth (cm)
St. 1	50 cm under water surface	7.97	8.79	20.90	150.00	25
	25% (total depth)	7.90	8.70	20.80	149.00	
	50% (total depth)	7.89	8.75	20.80	146.00	
	65% (total depth)	7.85	8.75	20.80	142.00	
	80% (total depth)	7.70	8.65	20.80	146.00	
St. 6	50 cm under water surface	7.85	8.57	17.20	126.00	30
	25% (total depth)	7.85	8.57	17.50	124.00	
	50% (total depth)	7.85	8.57	17.50	130.00	
	65% (total depth)	7.85	8.57	17.50	129.00	
	80% (total depth)	7.85	8.57	17.50	128.00	
St. 10	50 cm under water surface	8.30	8.50	17.80	16.00	110
	25% (total depth)	8.30	8.44	17.40	16.00	
	50% (total depth)	8.30	8.43	17.20	16.00	
	65% (total depth)	8.30	8.43	17.20	16.00	
	80% (total depth)	8.29	8.40	16.80	32.00	

Table A1.3: Laboratory analysis of different chemical and physical parameters of Lake Nubia at some selected stations during January 2006.

Station	Position	NH_4 (mg/L)	NO_3 (mg/L)	NO_2 (mg/L)	Ortho-P (mg/L)	Total-P (mg/L)	TDS (mg/L)
St. 1	50 cm under water surface	0.09	0.40	0.003	0.160	0.35	114.0
	25% (total depth)	0.11	0.40	0.003	0.170	0.36	114.0
	50% (total depth)	0.05	0.40	0.003	0.170	0.33	113.0
	65% (total depth)	0.09	0.30	0.003	0.210	0.34	115.0
	80% (total depth)	0.09	0.40	0.003	0.160	0.33	116.0
St. 6	50 cm under water surface	0.20	0.30	0.007	0.150	0.24	116.5
	25% (total depth)	0.12	0.40	0.010	0.210	0.30	116.5
	50% (total depth)	0.13	0.40	0.018	0.180	0.29	116.5
	65% (total depth)	0.10	0.40	0.007	0.160	0.28	116.5
	80% (total depth)	0.08	0.30	0.008	0.170	0.29	116.5
St. 10	50 cm under water surface	0.07	0.60	0.003	0.100	0.18	137.6
	25% (total depth)	0.01	0.20	0.003	0.090	0.18	138.2
	50% (total depth)	0.01	0.40	0.003	0.070	0.17	135.7
	65% (total depth)	0.01	0.50	0.003	0.100	0.18	135.7
	80% (total depth)	0.01	0.80	0.003	0.110	0.20	133.8

Table A1.4: Laboratory analysis of different biological parameters of Lake Nubia at some selected stations during January 2006.

Station	Position	**Fecal coliform** (col./100 mL)			**Chl-a** (µg/L)	
		East	*Middle*	*West*	*Light*	*Dark*
St. 1	50 cm under water surface	10	10	10	11	
	50% (total depth)	5	10	10		6.2
	80% (total depth)	10	5	10		
St. 6	50 cm under water surface	5	-	-	8	
	50% (total depth)	5	10	-		7.5
	80% (total depth)	-	-	-		
St. 10	50 cm under water surface	-	-	-	30	
	50% (total depth)	-	-	-		29
	80% (total depth)	-	-	-		

Table A1.5: Total suspended solids (TSS) concentrations of Lake Nubia at some selected stations during January 2006.

Station	Position	**TSS** (mg/L)		
		East	*Middle*	*West*
St. 1	50 cm under water surface	147	140	145
	25% (total depth)	149	138	154
	50% (total depth)	149	140	146
	65% (total depth)	144	135	144
	80% (total depth)	146	140	145
St. 6	50 cm under water surface	120	120	126
	25% (total depth)	120	119	126
	50% (total depth)	121	125	134
	65% (total depth)	121	125	134
	80% (total depth)	124	122	133
St. 10	50 cm under water surface	23	15	12
	25% (total depth)	23	15	12
	50% (total depth)	23	14	8
	65% (total depth)	25	13	5
	80% (total depth)	32	16	7

A1.2 Measured data during verification period (February 2007)

Table A1.6: Field information of Lake Nubia at some selected stations during February 2007.

Site name	Station number	Distance upstream AHD (km)	Date of sampling	Time of sampling	Total depth (m)	Air temperature (°C)
ElDaka (23)	St. 1	487.5	2nd	8.35	9.00	21.00
Kagnarty (6)	St. 6	394.0	8th	7.45	8.00	19.00
Dabarosa (22)	St. 10	337.5	14th	9.20	25.00	18.00

Table A1.7: Field measurments of different chemical and physical parameters of Lake Nubia at some selected stations during February 2007.

Station	Position	pH	DO (mg/L)	Water Temperature (°C)	Turbidity (NTU)	Secchi depth (cm)
St. 1	50 cm under water surface	7.79	9.45	17.0	92	25
	25% (total depth)	7.78	9.46	17.1	96	
	50% (total depth)	7.79	9.44	16.9	97	
	65% (total depth)	7.76	9.46	16.9	94	
	80% (total depth)	7.80	9.37	17.0	96	
St. 6	50 cm under water surface	7.89	8.80	16.8	71	30
	25% (total depth)	7.87	8.75	16.7	75	
	50% (total depth)	7.85	8.70	16.5	75	
	65% (total depth)	7.85	8.73	16.3	75	
	80% (total depth)	7.76	8.84	16.3	81	
St. 10	50 cm under water surface	8.44	9.92	17.0	8	180
	25% (total depth)	8.44	9.90	16.8	11	
	50% (total depth)	8.42	9.85	16.9	9	
	65% (total depth)	8.29	9.80	17.0	10	
	80% (total depth)	8.28	9.74	17.0	12	

Table A1.8: Laboratory analysis of different chemical and physical parameters of Lake Nubia at some selected stations during February 2007.

Station	Position	NH_4 (mg/L)	NO_3 (mg/L)	NO_2 (mg/L)	Ortho-P (mg/L)	TDS (mg/L)
St. 1	50 cm under water surface	<0.01	0.40	0.004	0.15	114
	25% (total depth)	<0.01	0.36	0.004	0.15	114
	50% (total depth)	<0.01	0.33	0.004	0.15	114
	65% (total depth)	<0.01	0.32	0.004	0.15	114
	80% (total depth)	<0.01	0.32	0.002	0.15	114
St. 6	50 cm under water surface	<0.01	0.20	0.002	0.110	112
	25% (total depth)	<0.01	0.20	0.001	0.110	112
	50% (total depth)	<0.01	0.20	0.003	0.120	111
	65% (total depth)	<0.01	0.20	0.003	0.110	112
	80% (total depth)	<0.01	0.20	0.003	0.110	112
St. 10	50 cm under water surface	<0.01	0.66	0.018	0.10	125
	25% (total depth)	<0.01	0.66	0.012	0.12	124
	50% (total depth)	<0.01	0.67	0.010	0.13	124
	65% (total depth)	<0.01	0.67	0.010	0.15	125
	80% (total depth)	<0.01	0.65	0.010	0.15	123

Table A1.9: Laboratory analysis of different biological parameters of Lake Nubia at some selected stations during February 2007.

Station	Position	**Fecal coliform** (col./100 mL)			**Chl-a** (µg/L)	
		East	*Middle*	*West*	*Light*	*Dark*
St. 1	50 cm under water surface	10	10	10	15.8	
	50% (total depth)	-	5	10		14.8
	80% (total depth)	-	10	15		
St. 6	50 cm under water surface	5	-	10	22.9	
	50% (total depth)	-	5	5		21.1
	80% (total depth)	5	5	-		
St. 10	50 cm under water surface	10	20	10	35.5	
	50% (total depth)	-	-	-		29.4
	80% (total depth)	-	-	10		

Table A1.10: Total suspended solids (TSS) concentrations of Lake Nubia at some selected stations during February 2007.

Station	Position	**TSS** (mg/L)		
		East	*Middle*	*West*
St. 1	50 cm under water surface	80	98	91
	25% (total depth)	87	100	93
	50% (total depth)	87	102	97
	65% (total depth)	94	99	95
	80% (total depth)	93	98	98
St. 6	50 cm under water surface	95	84	102
	25% (total depth)	98	87	107
	50% (total depth)	91	88	95
	65% (total depth)	95	90	96
	80% (total depth)	91	95	97
St. 10	50 cm under water surface	20	10	9
	25% (total depth)	22	13	12
	50% (total depth)	17	11	8
	65% (total depth)	24	13	8
	80% (total depth)	25	15	12

Appendix 2: Lake Nubia model: water quality results of calibration and verification processes

A2.1 Water quality results of calibration process (January 2006)

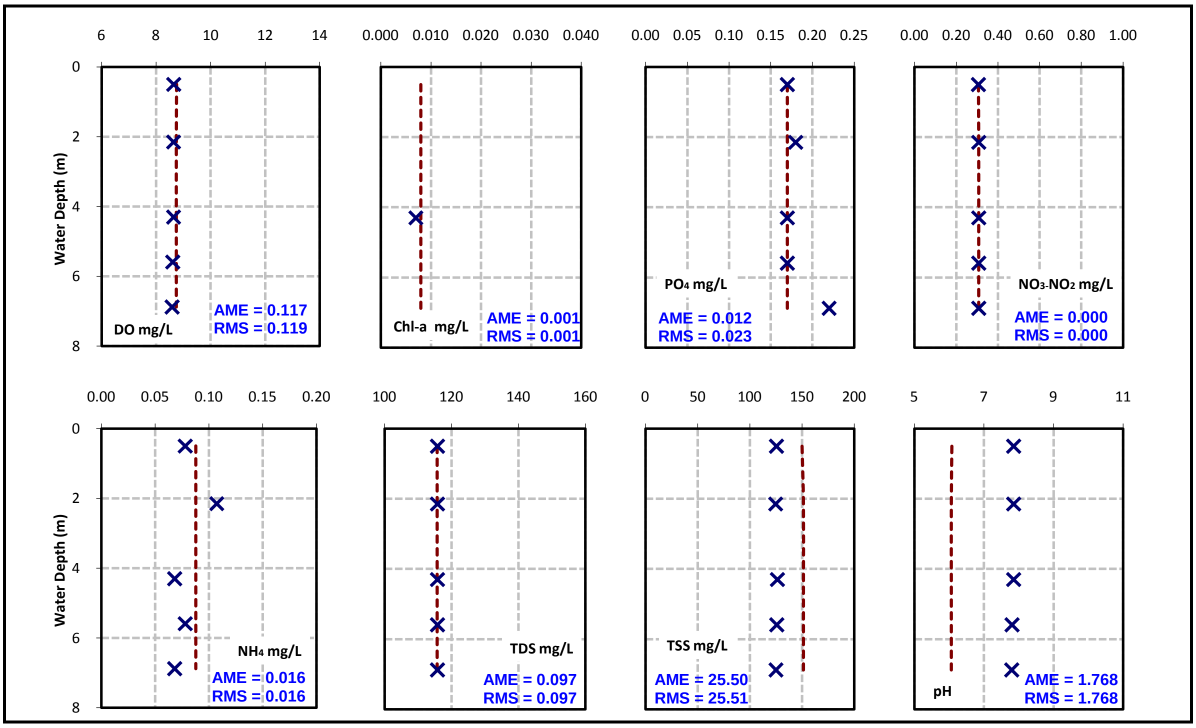

Figure A2.1: Model calibration: vertical profiles of measured (x) and simulated (---) key water quality parameters in Lake Nubia at St. 3 on 9. January 2006.

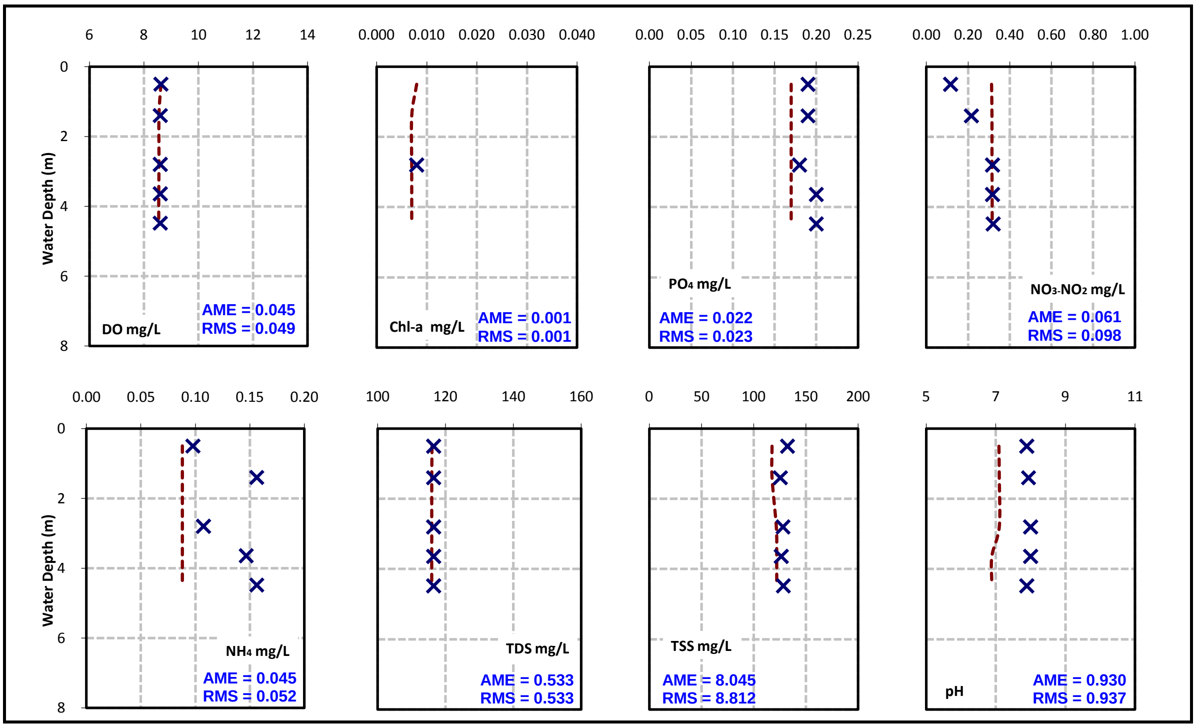

Figure A2.2: Model calibration: vertical profiles of measured (x) and simulated (---) key water quality parameters in Lake Nubia at St. 5 on 11. January 2006.

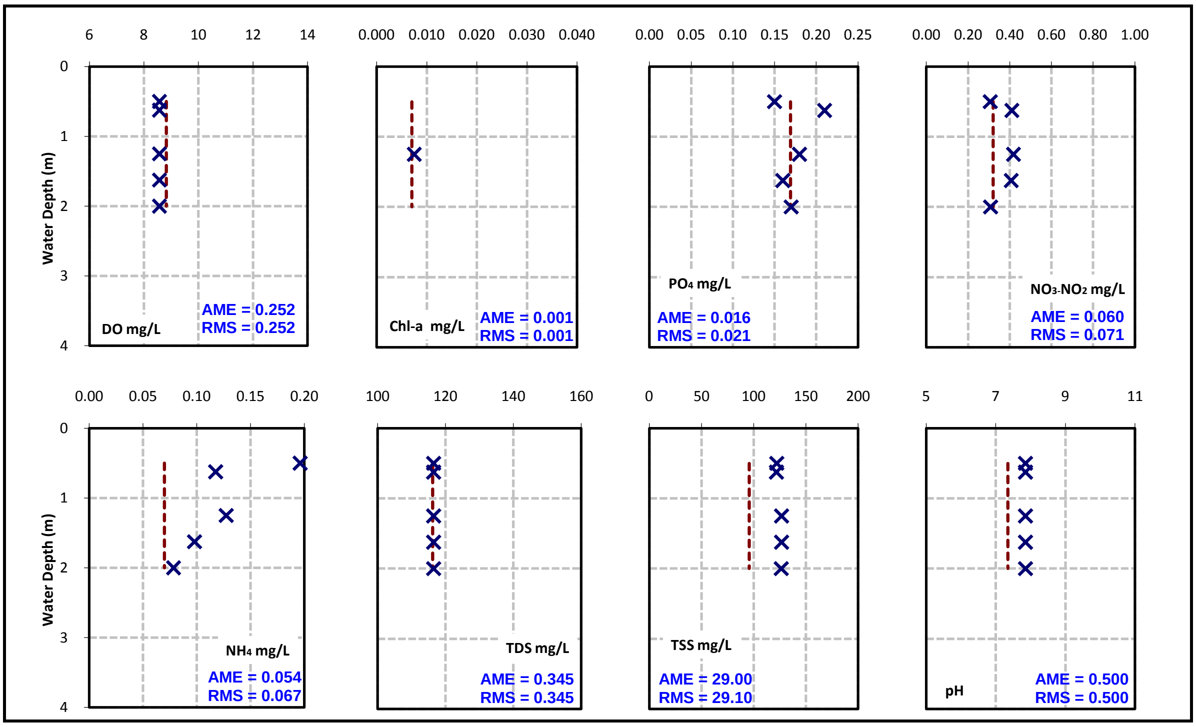

Figure A2.3: Model calibration: vertical profiles of measured (x) and simulated (---) key water quality parameters in Lake Nubia at St. 6 on 13. January 2006.

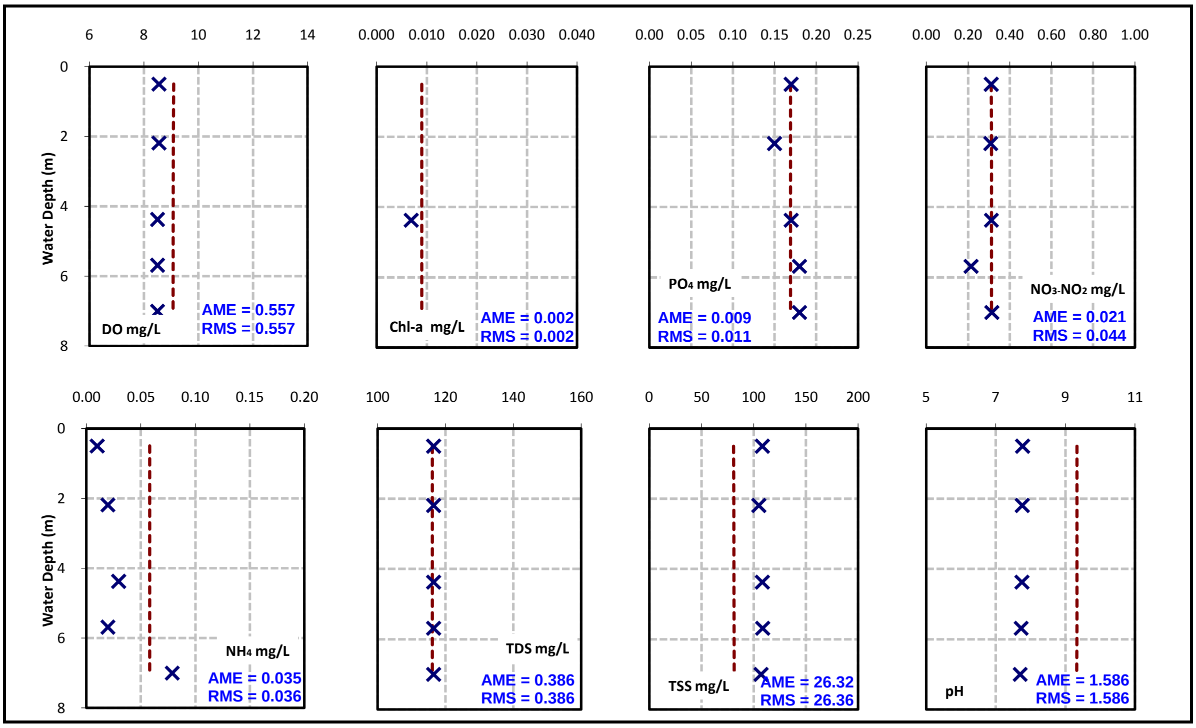

Figure A2.4: Model calibration: vertical profiles of measured (x) and simulated (---) key water quality parameters in Lake Nubia at St. 7 on 14. January 2006.

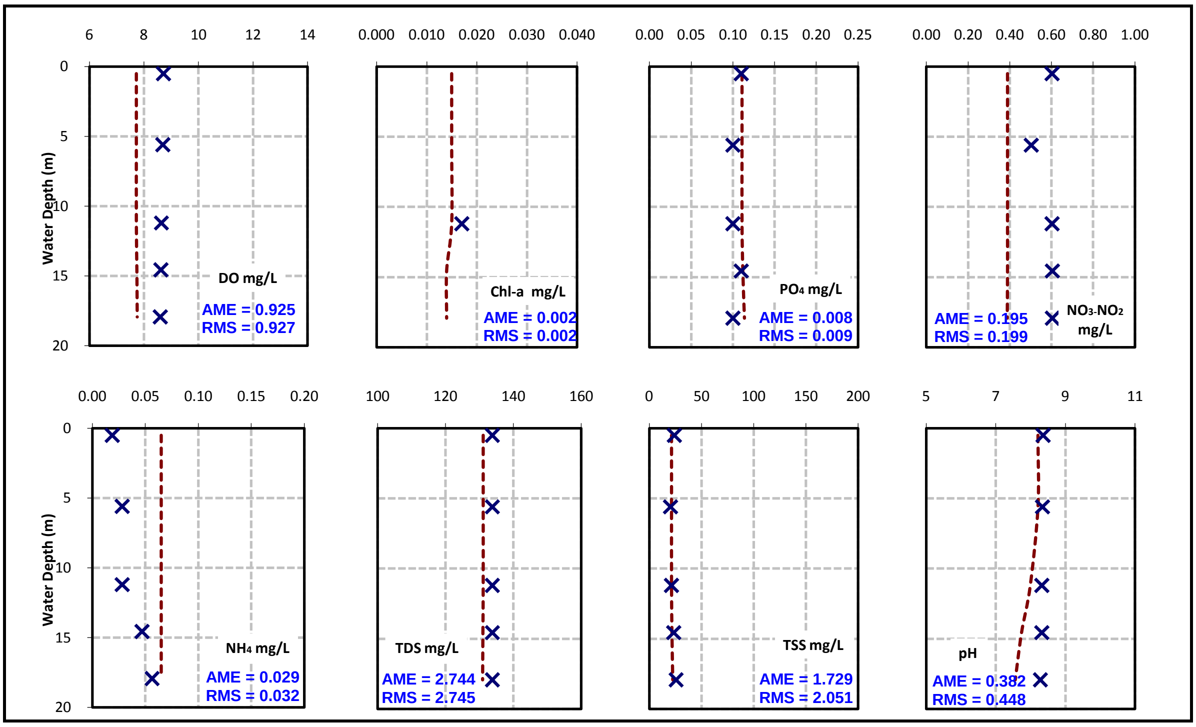

Figure A2.5: Model calibration: vertical profiles of measured (x) and simulated (---) key water quality parameters in Lake Nubia at St. 9 on 18. January 2006.

A2.2 Water quality results of verification process (February 2007)

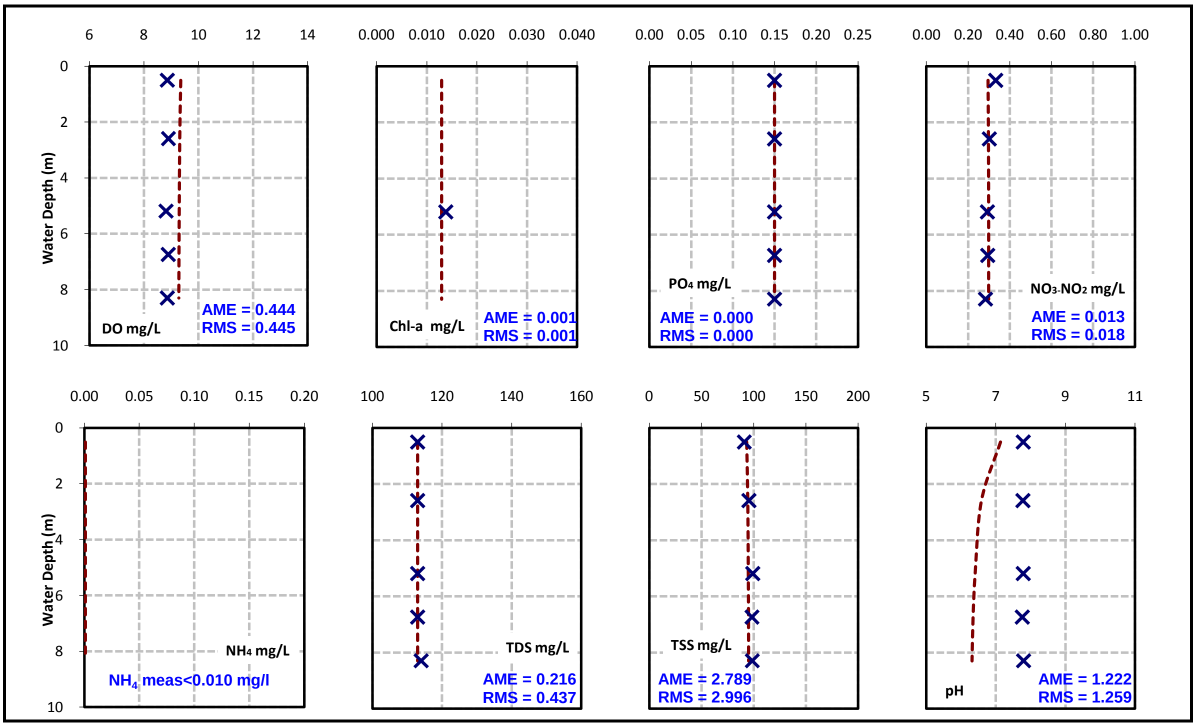

Figure A2.6: Model verification: vertical profiles of measured (x) and simulated (---) key water quality parameters in Lake Nubia at St. 2 on 2. February 2007.

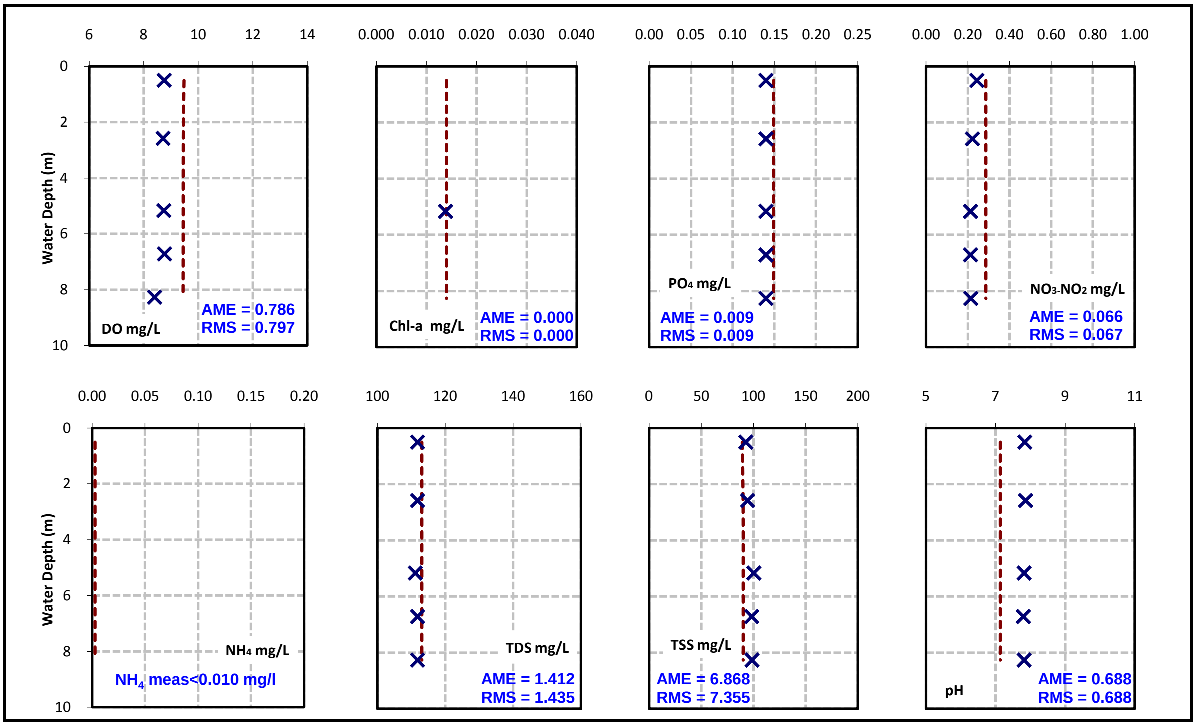

Figure A2.7: Model verification: vertical profiles of measured (x) and simulated (---)key water quality parameters in Lake Nubia at St. 3 on 4. February 2007.

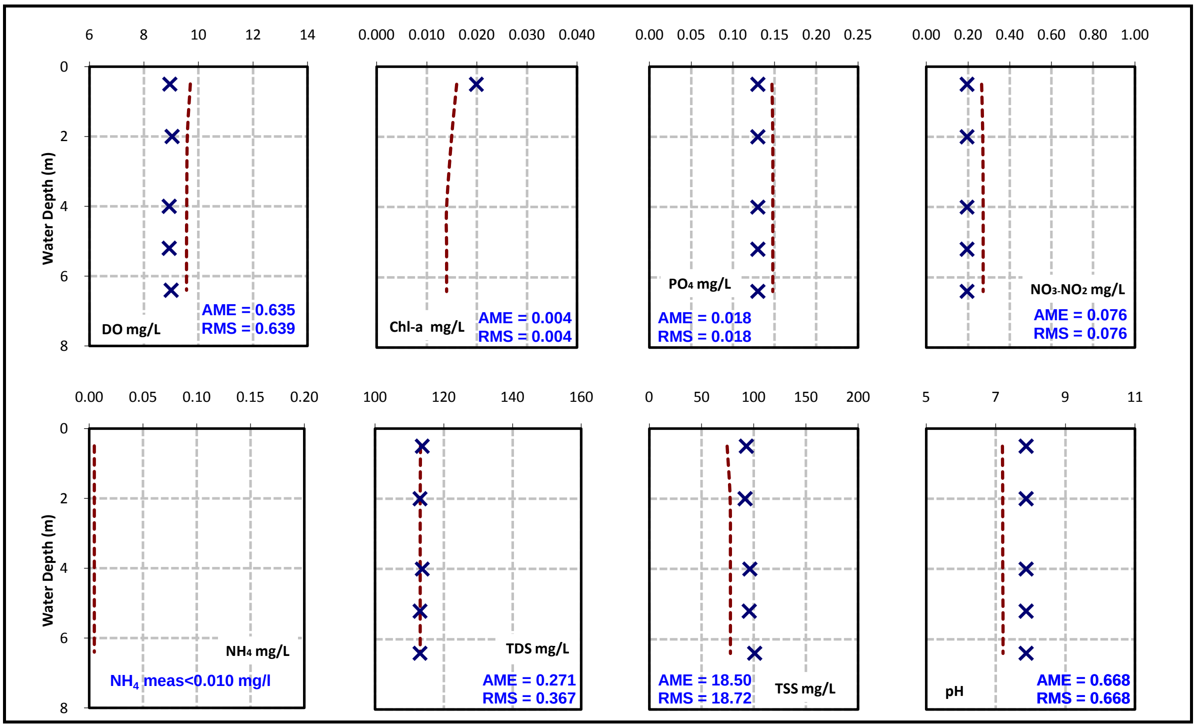

Figure A2.8: Model verification: vertical profiles of measured (x) and simulated (---)key water quality parameters in Lake Nubia at St. 5 on 5. February 2007.

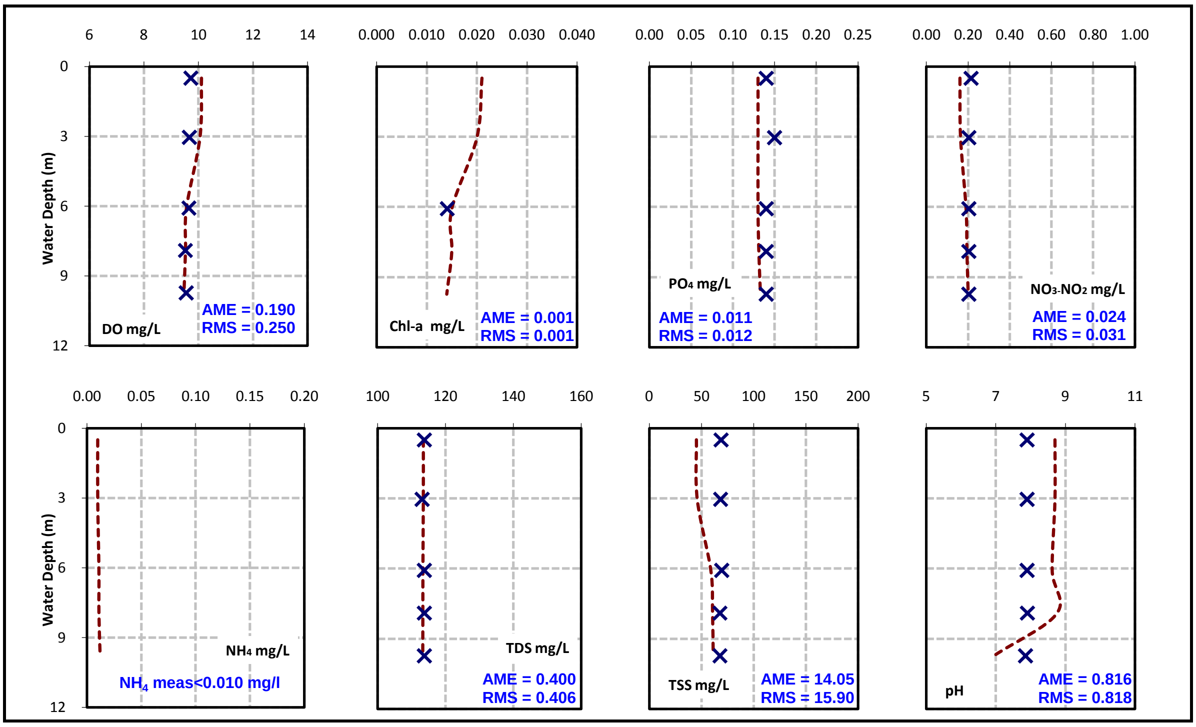

Figure A2.9: Model verification: vertical profiles of measured (x) and simulated (---)key water quality parameters in Lake Nubia at St. 7 on 9. February 2007.

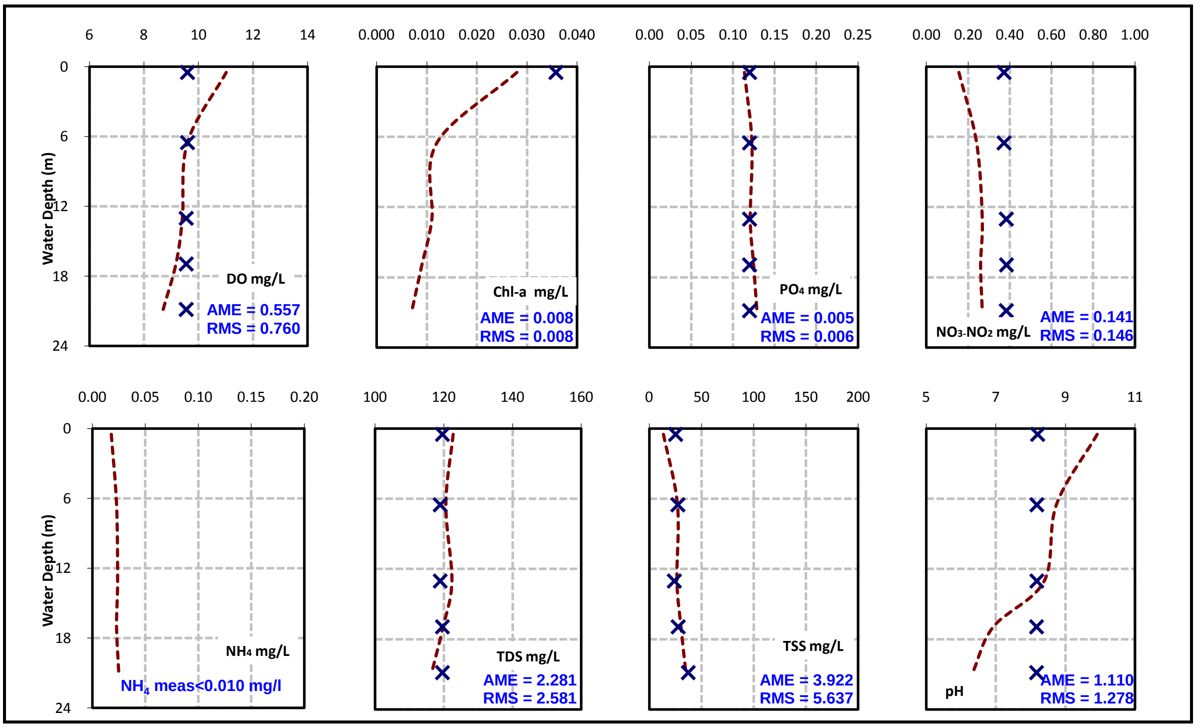

Figure A2.10: Model verification: vertical profiles of measured (x) and simulated (---)key water quality parameters in Lake Nubia at St. 9 on 13. February 2007.

Appendix 3: NSF water quality index

A3.1 Q-value curves of NSF WQI

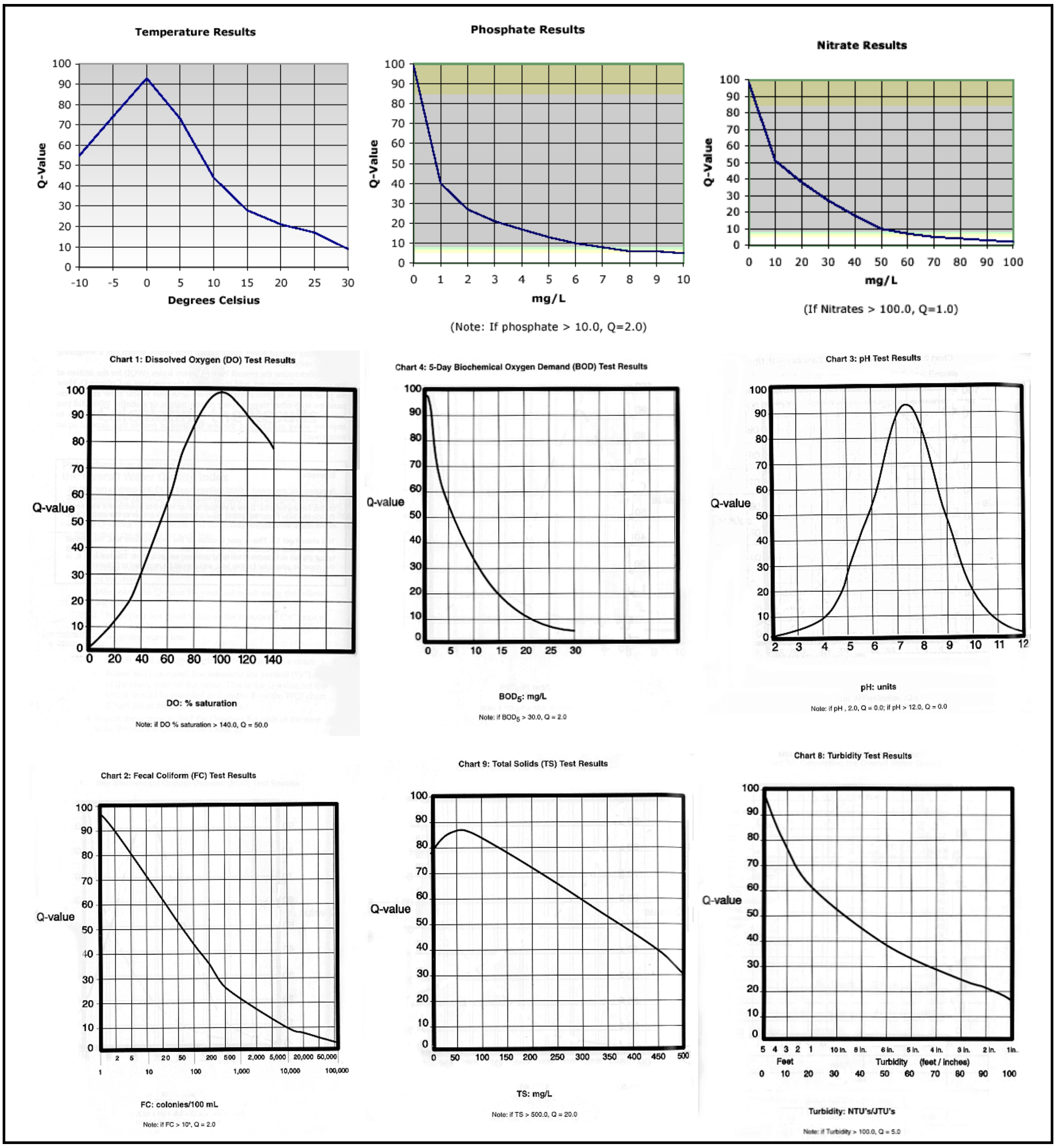

Figure A3.1: Q-value curves of NSF WQI for different parameters (Wilkes University website).

A3.2 NSF WQI sample calculations for Lake Nubia

Table A3.1: NSF WQI for Lake Nubia at different stations (for measured parameters, January 2006).

Stations	T_w (°C) Reference	T_w (°C)	T_w change (°C)	Q –T_w	NO_3-NO_2 (mg/L)	Q - NO_3
3	20.82	19.76	1.060	90	0.266	97
6	20.82	17.44	3.380	80	0.370	97
9	20.82	15.94	4.880	74	0.583	96

Stations	DO (mg/L)	DO %SAT	Q - DO	TSS (mg/L)	Q - TSS
3	8.63	94.418	98	125.40	81
6	8.57	89.486	95	124.67	81
9	8.65	87.565	93	22.93	84

Stations	pH	Q - pH	PO_4 (mg/L)	Q - PO_4	FC (N/100 mL)	Q - FC
3	7.83	89	0.182	93	17	65
6	7.85	89	0.174	93	3	86
9	8.32	73	0.104	96	0	99

Stations	TUR (NTU)	Q - TUR	NSF WQI	Quality rating
3	129.40	5	**79.69**	**Good**
6	127.40	5	**81.76**	**Good**
9	28.80	54	**85.93**	**Good**

Appendix 4: Results of climate change study

A4.1 Hydrodynamic results (thermal structure)

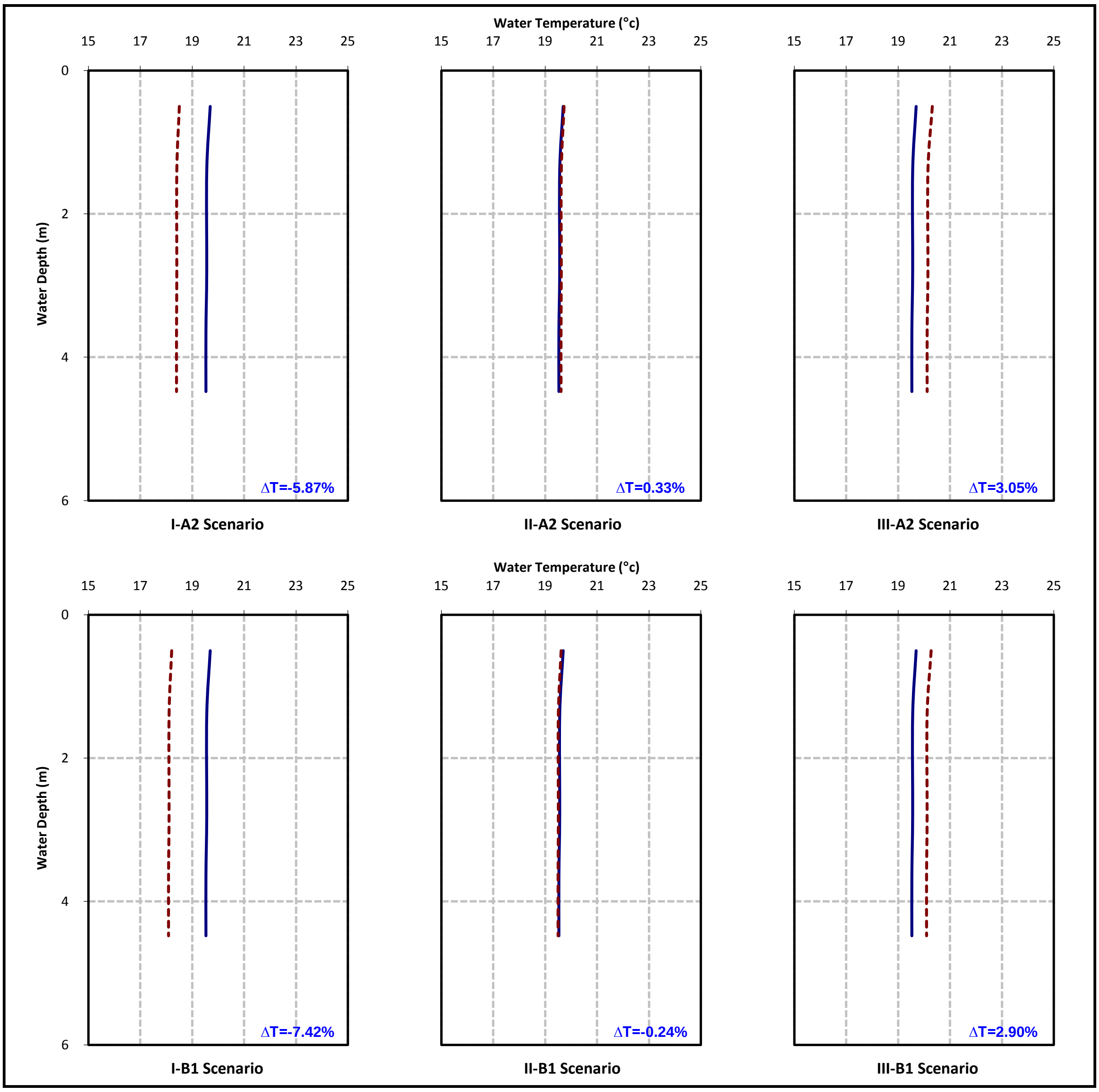

Figure A4.1 : Lake Nubia water temperature profiles at St. 5 due to global climate change [- - -] for different scenarios, compared with the simulated base case [—].

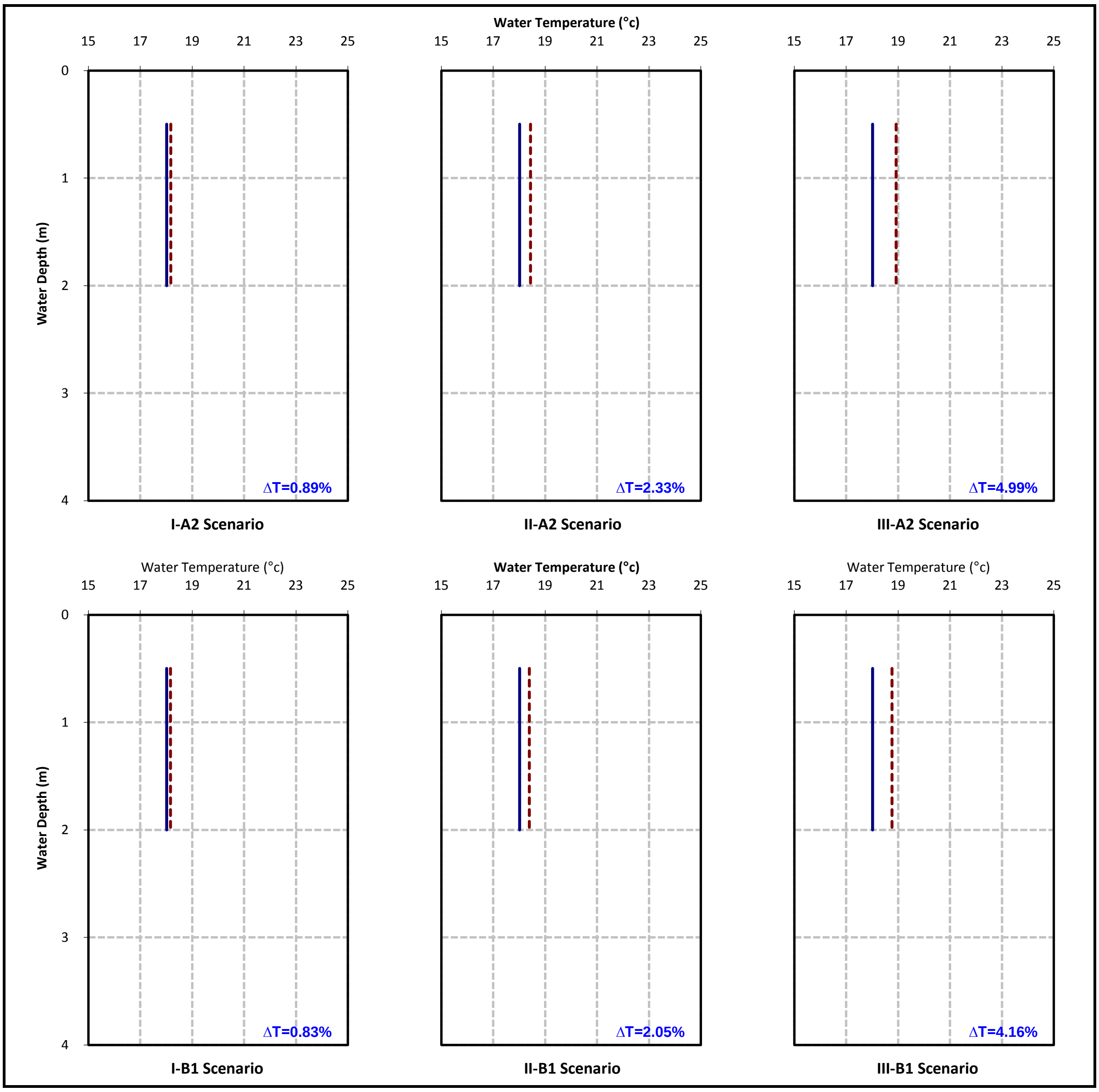

Figure A4.2 : Lake Nubia water temperature profiles at St. 6 due to global climate change [- - -] for different scenarios, compared with the simulated base case [—].

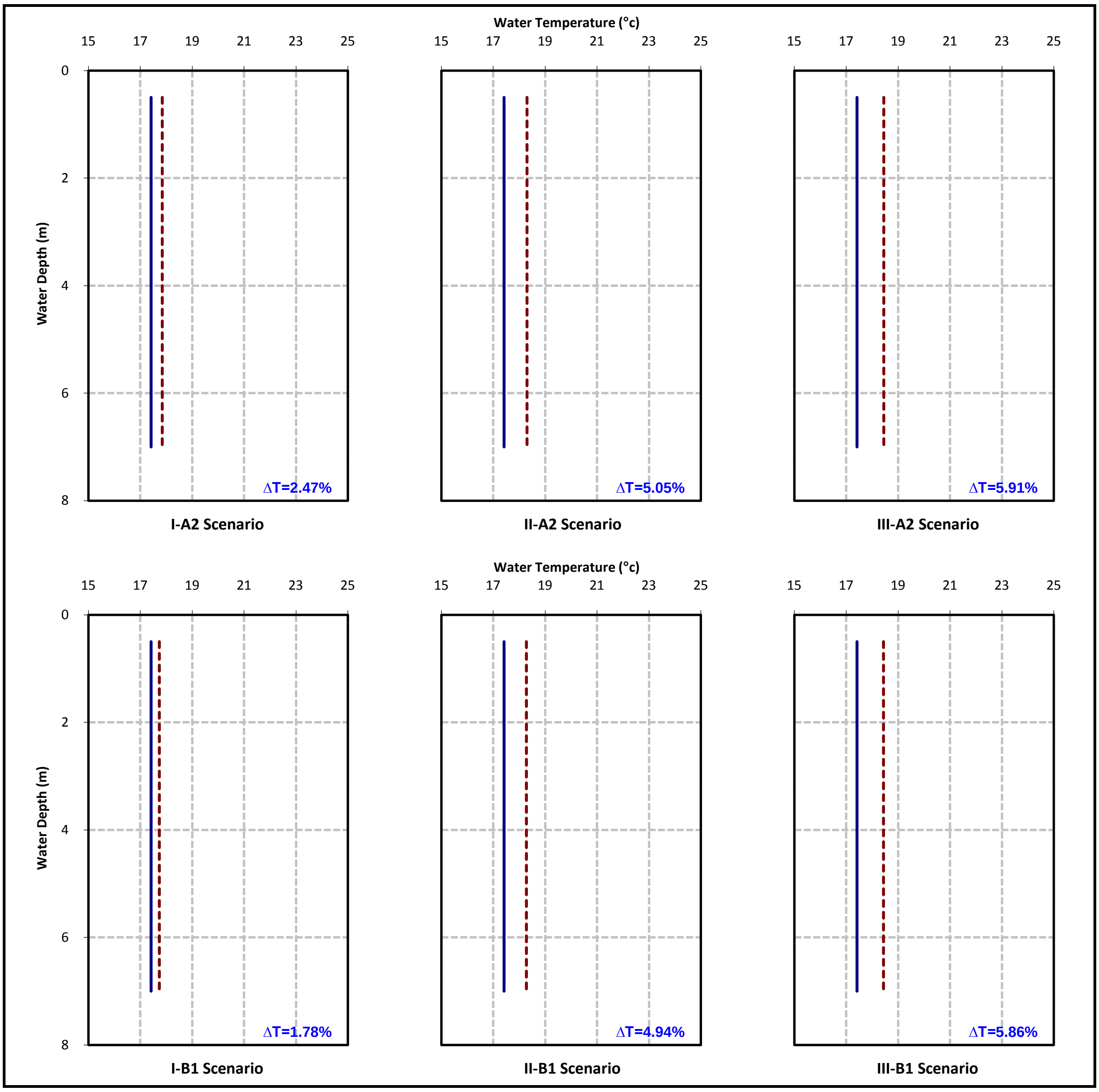

Figure A4.3 : Lake Nubia water temperature profiles at St. 7 due to global climate change [- - -] for different scenarios, compared with the simulated base case [—].

A4.2 Water quality results

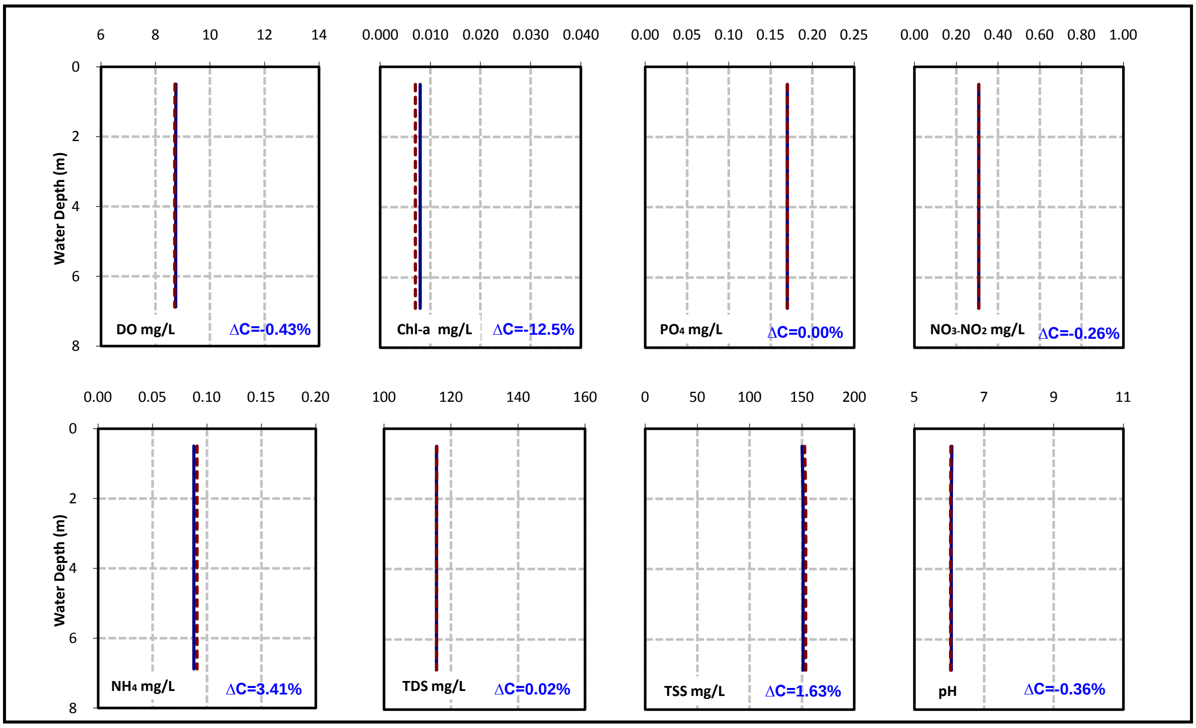

Figure A4.4: Lake Nubia water quality characteristics profiles due to global climate change [- - -] for I-A2 scenario at St. 3, compared with the simulated base case [—].

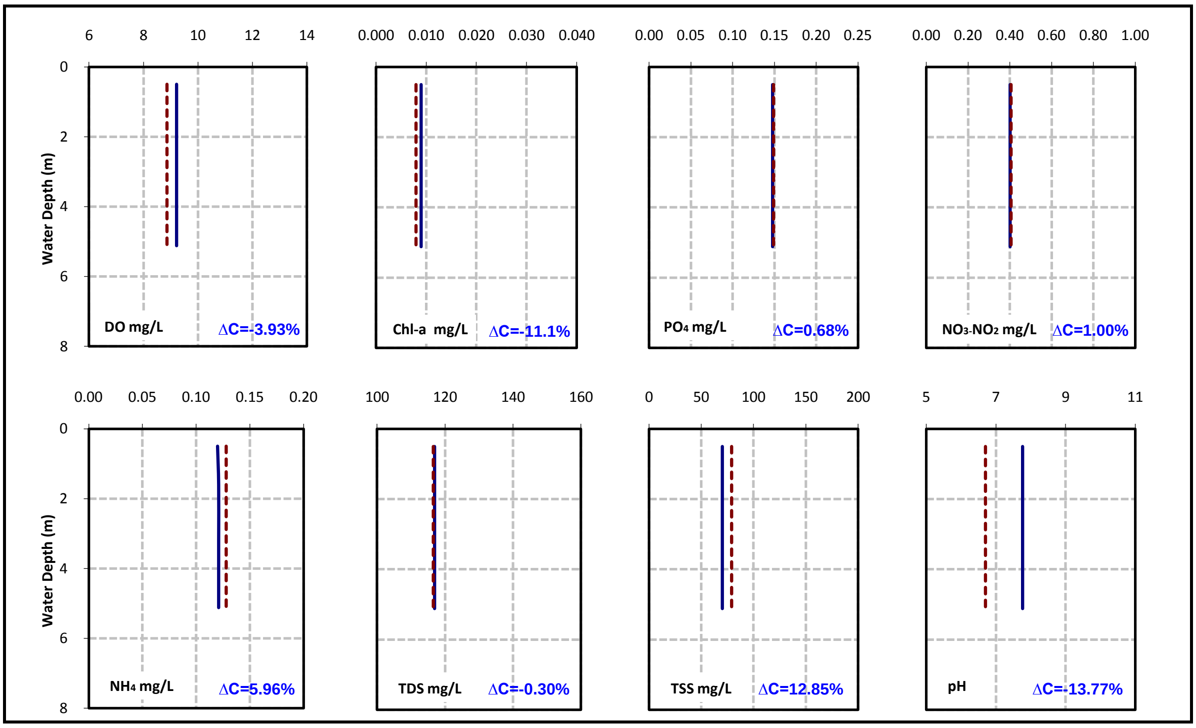

Figure A4.5: Lake Nubia water quality characteristics profiles due to global climate change [- - -] for I-B1 scenario at St. 8, compared with the simulated base case [—].

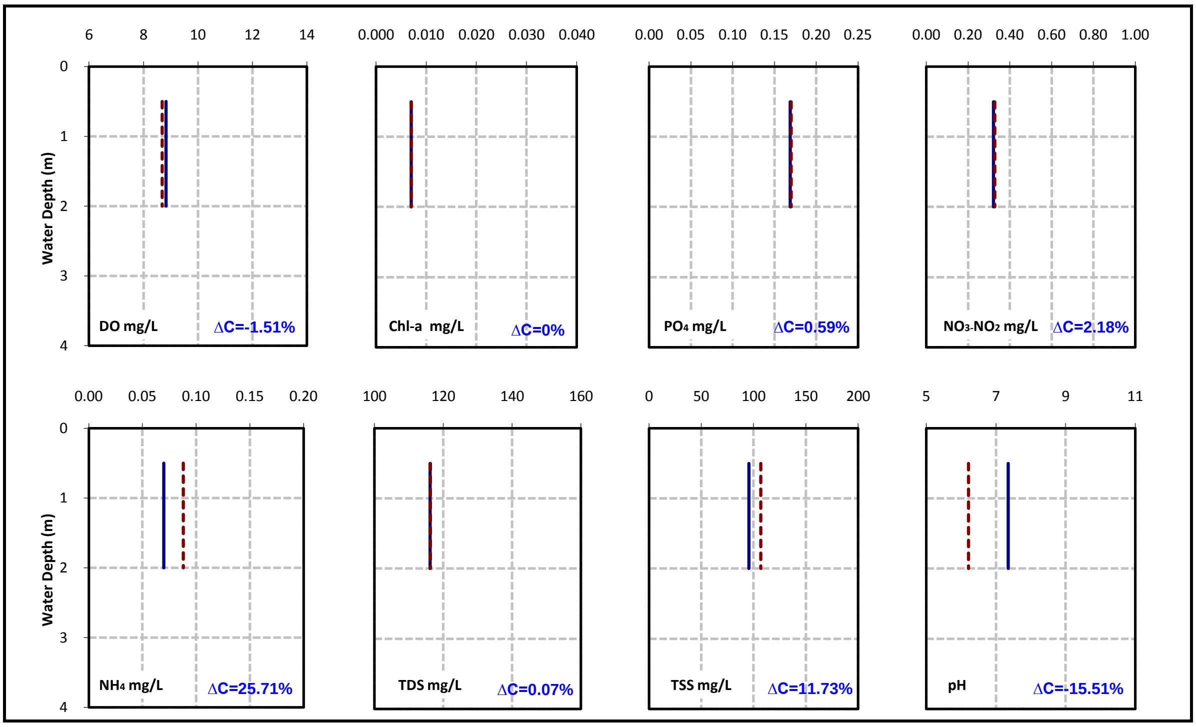

Figure A4.6: Lake Nubia water quality characteristics profiles due to global climate change [- - -] for II-A2 scenario at St. 6, compared with the simulated base case [—].

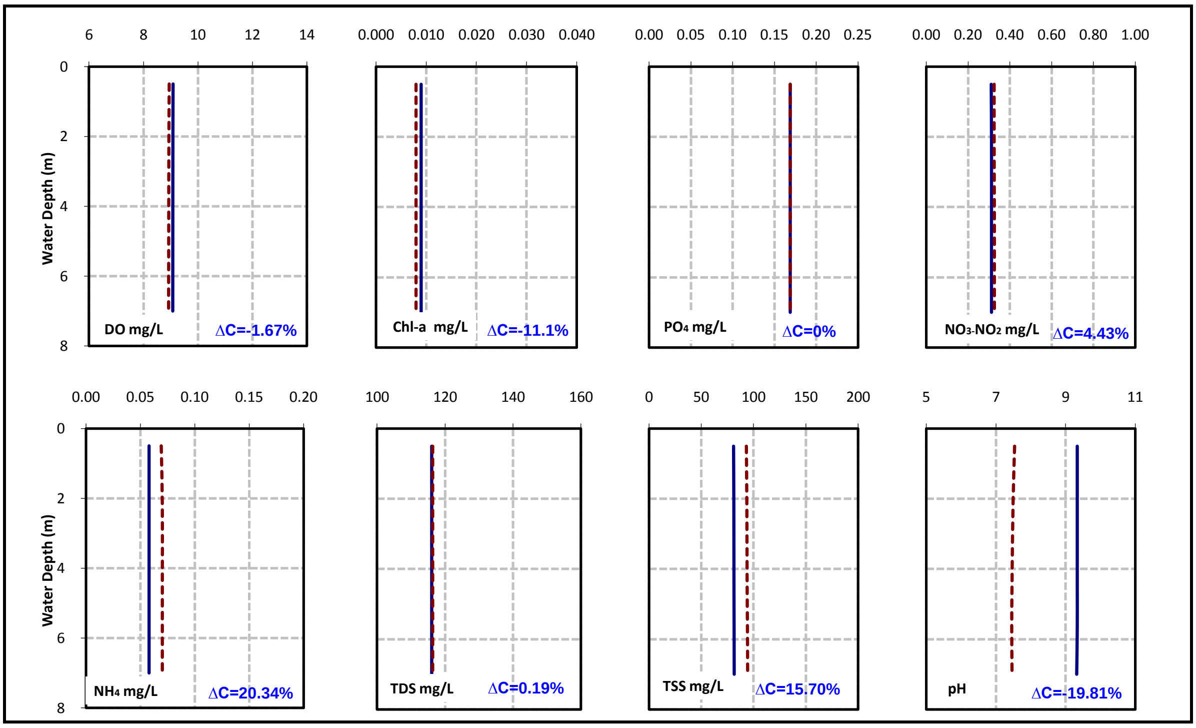

Figure A4.7: Lake Nubia water quality characteristics profiles due to global climate change [- - -] for II-B1 scenario at St. 7, compared with the simulated base case [—].

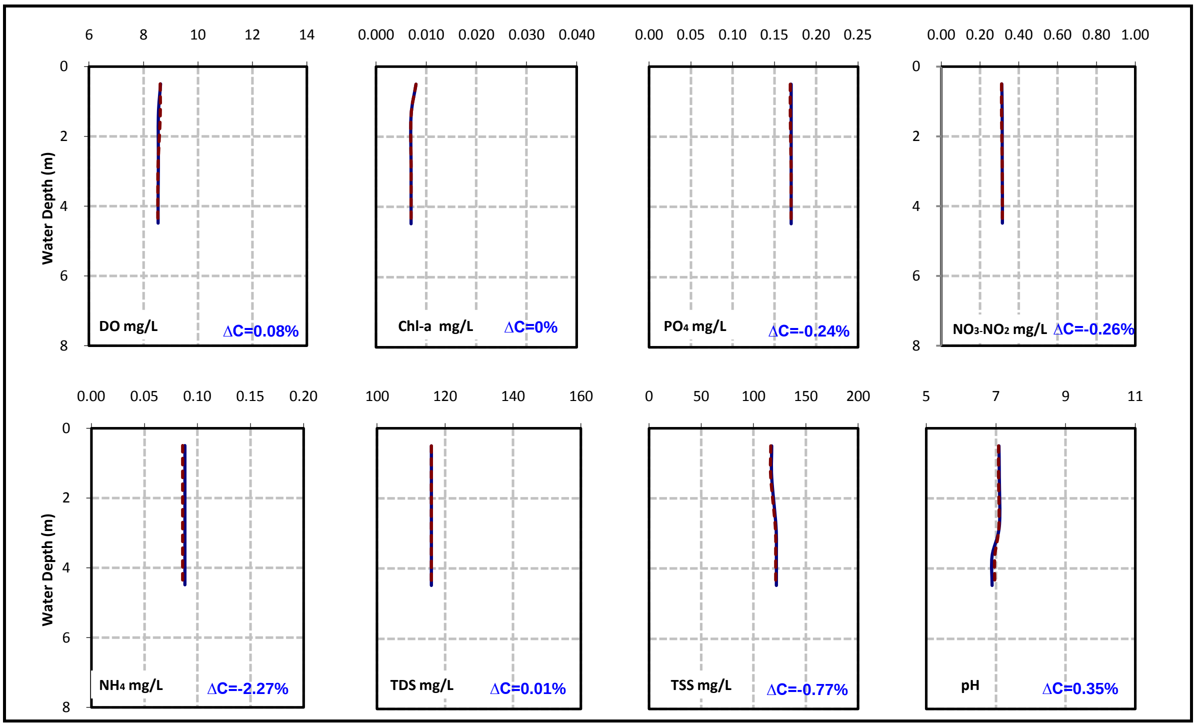

Figure A4.8: Lake Nubia water quality characteristics profiles due to global climate change [- - -] for III-A2 scenario at St. 5, compared with the simulated base case [—].

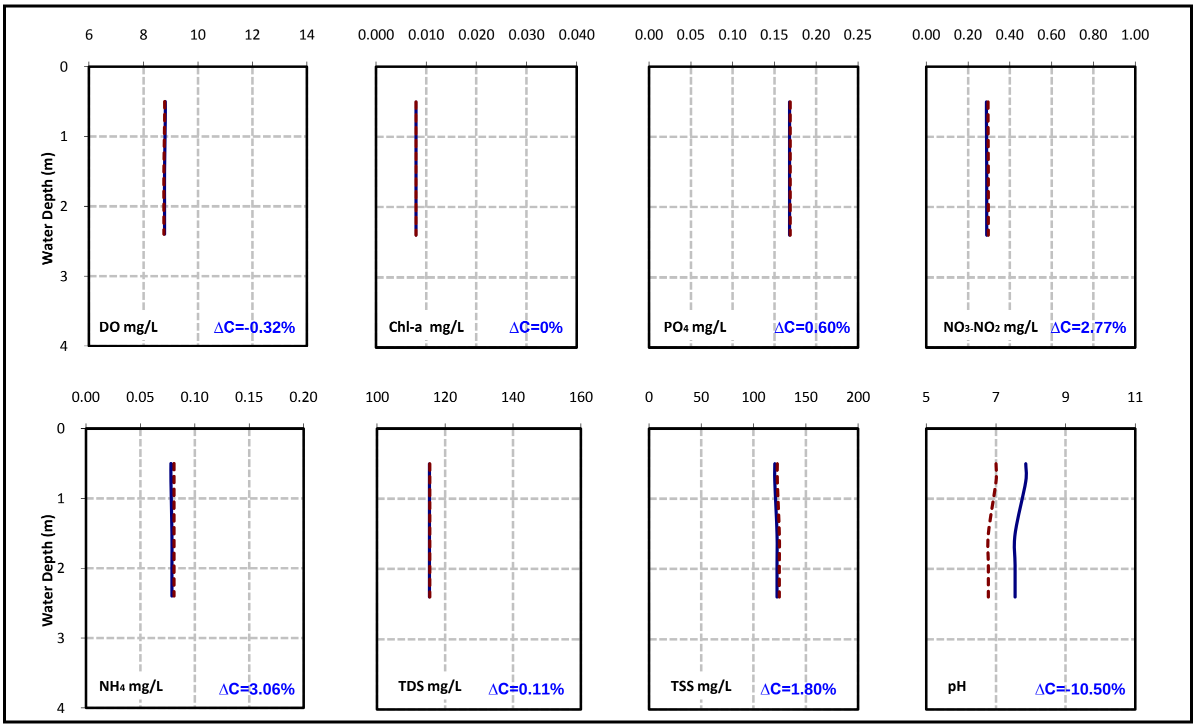

Figure A4.9: Lake Nubia water quality characteristics profiles due to global climate change [- - -] for III-B1 scenario at St. 4, compared with the simulated base case [—].